Volume

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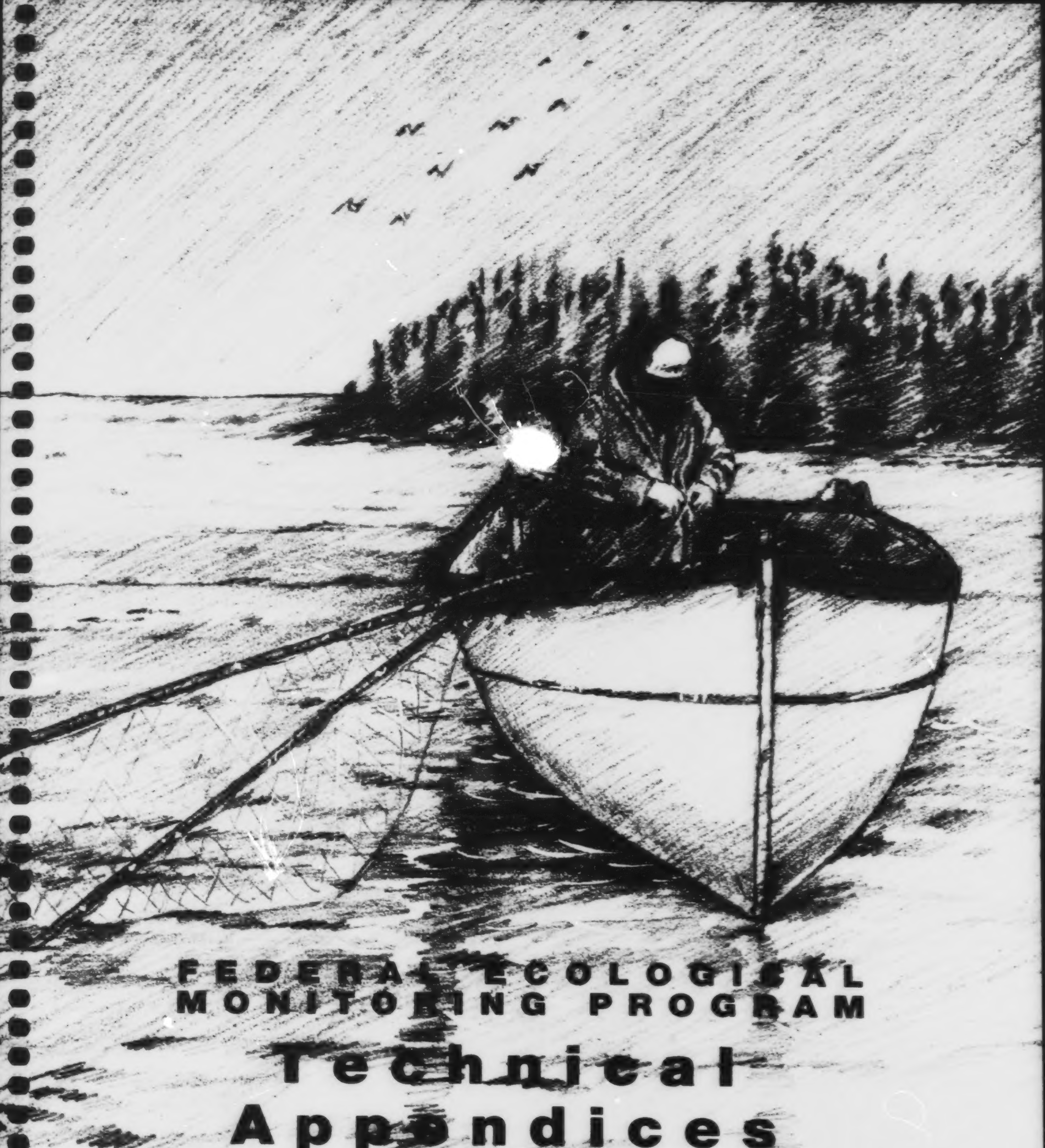

FEDERAL ECOLOGICAL MONITORING PROGRAM

Technical Appendices

Environment Canada
Environnement Canada

Fisheries and Oceans
Pêches et Océans

FEMP TECHNICAL REPORTS

A large number of technical reports were produced under the auspices of FEMP. Most of these reports have already been published; some as part of the FEMP report series, some as part of the DFO technical report series, and some as DOE reports. In addition, there are still some reports that are in preparation. All of these reports are listed in the the following pages to provide the reader with a complete listing of FEMP sponsored studies.

The reports are listed in alphabetical order. The designation of [DOE] or [DFO] after each listing indicates the availability of each report.

For information on DOE designated reports contact

FEMP Reports
Water Resources Branch
Environment Canada
513 - 269 Main Street
Winnipeg, Manitoba R3C 1B2

For information on DFO designated reports contact

FEMP Reports
Department of Fisheries and Oceans
Freshwater Institute
501 University Crescent
Winnipeg, Manitoba R3T 2N6

The five reports published here have not been published previously.

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SPECIAL NOTE

Conclusions, recommendations or opinions expressed in reports prepared for the Federal Ecological Monitoring Program are those of the authors and do not necessarily represent those of the sponsoring departments.

Federal Ecological Monitoring Program
Final Water Quality Report

Prepared For:

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Western and Northern Region
Water Quality Branch
Regina, Saskatchewan

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Executive Summary

Environment Canada and the Department of Fisheries and Oceans undertook in 1986 a five year program of ecological monitoring in northern Manitoba which has come to be known as the Federal Ecological Monitoring Program (FEMP). The overall purpose of this program is to address Canada's obligations under the Northern Flood Agreement (NFA) with respect to requirements for ecological monitoring and the identification and disclosure of adverse effects of Churchill River diversion (CRD) and Lake Winnipeg regulation (LWR). This report presents the results of the water quality monitoring program implemented by Environment Canada as part of the FEMP. Current (1987-1989) water quality is described, statistically significant changes in physical and chemical water quality since the LWCNR project began operation are documented and, where possible, the cause of these changes is determined. Any adverse effects directly resulting from changes in water quality caused by the Project are identified. The definition of an adverse effect employed in this report is a statistically significant change in water quality, caused by the hydroelectric developments, which has a detrimental effect on one or more of the traditional water uses of the communities. A detrimental effect is a change in one or more water quality variables such that the variable is not within the acceptable limits of one or more of the Canadian Water Quality Guidelines for drinking water, recreational use, or protection of aquatic life.

Current water quality was documented in a three year study (January 1987 to October 1989) of 61 physical and chemical water quality parameters at 11 sites on the Rat, Burntwood, Footprint, upper and lower Nelson, Jack, and Aiken rivers. The hydrologic regimes at 4 of the sites were modified by CRD (Rat R. above Threepoint L.; the Burntwood R. at the outlet of Threepoint L., at Thompson, and at the inlet to Split Lake), while 3 sites were affected by LWR (Nelson R. below Sea River Falls and below the confluence with the Grass R.; the Jack R. (west channel) at Norway House). A site on Nelson R. at the outlet to Split Lake was affected by both projects. The remaining three sites were on unregulated rivers (Burntwood R. above Threepoint L., Footprint R. above Footprint L., Aiken R.) to provide a measure of current water quality in natural rivers in the study area and, where sufficient historical data existed, to determine the degree of natural variation in water quality between the pre- and post-development study periods. Characteristics of water quality examined in the FEMP study included the annual mean values of the physical and chemical variables, their range of variation over the 3 years of study, differences between the ice-free and ice-cover seasons, and the relationship, if any, with river discharge. These data were compared with all available data for the pre-development, construction, and immediate post-development periods to determine if significant changes had occurred and if these changes can be attributed to LWR, CRD, or to both projects.

A study of the bacteriological quality of recreational waters in and around the communities of Cross Lake, Norway House, Split Lake, York Landing, and Nelson House also was undertaken to evaluate the current quality of waters for recreational (principally swimming) uses. The study did not examine the effect of the hydroelectric projects on bacterial water quality.

Current Water Quality and Compliance with Canadian Guidelines

Physical and chemical water quality at all 11 sites generally meets the Canadian Water Quality Guidelines for drinking water, recreational uses, and protection of aquatic life although there are a number of exceptions. Mean turbidity exceeds the Canadian drinking water guideline and mean extractable Al and total Cu exceed the aquatic life guidelines at all sites. At the natural river sites and on the upper Nelson River (i.e., above Split Lake), mean turbidity is in the range of 6 to 19 NTU. Turbidity on the CRD between Threepoint and Split lakes is higher, ranging from 20 to 35 NTU, due to shoreline and bank erosion caused by the diversion. Levels of total Pb, Cd, and Zn occasionally exceed the respective aquatic life guidelines in the natural rivers and at sites on the CRD, but never in the Jack River or in the Nelson River above Split Lake.

The natural rivers and sites on the CRD are distinguished by mean extractable Fe exceeding the drinking water and aquatic life guidelines. Mean extractable Fe is below the drinking water and aquatic life guidelines in the Nelson River at sites above Split

Lake and in the Jack River. Both guidelines are exceeded in the Nelson River at the outlet of Split Lake. The high Fe levels in Split Lake are a direct result of CRD.

The consistent occurrence of extractable Al and total Cu concentrations in excess of the aquatic life guidelines, and of extractable Fe levels above both the aquatic life and drinking water guidelines in the natural rivers, suggests a review of the applicability of these guidelines in the study area is in order.

Bacterial Studies

The studies of fecal coliform levels in recreational waters indicate contamination is not a significant concern. No clear evidence of contamination was found at any site, including the vicinity of sewage treatment plant outfalls where contamination might be expected. However, the limitations of the sampling program with respect to the number and frequency of samples prevent conclusion that fecal coliform levels comply with national water quality guidelines.

Changes in Water Quality Related to the Hydroelectric Developments

Norway House Area

Impacts of LWR on water quality in the Nelson River below Sea River Falls and in the Jack River could not be assessed due to limited (Sea River Falls) or non-existent (Jack River) pre-development data records.

Nelson House and Threepoint Lake Area

The only adverse effect of CRD on water quality in the Rat River above Threepoint Lake was an approximately 3-fold increase in turbidity between the 1972-1973 pre-development study period and the 1987-1989 FEMP study. This increase was caused primarily by bank and shoreline erosion in the southern part of Notigi Reservoir.

Many other significant changes also occurred, although none can be classified as adverse. Annual mean conductivity, alkalinity (total), hardness (total), TOC, ortho PO_4, Ca, and reactive Si decreased by 15 to 60%, while Na, K, Cl, and F, increased by 10-80%. These changes were a reflection of the differences in water quality between the Churchill River and the pre-existing Rat River. In most cases, the changes were no greater than the difference between the Rat and Churchill rivers. The only exception was TOC, in which the decrease in concentration was greater than the difference between the rivers. This suggests some loss of TOC is occurring as the Churchill River passes through Southern Indian Lake and Notigi Reservoir. The total P concentration in the Rat River did not change following diversion despite a lower concentration in the Churchill River. Release of P from flooded soils and vegetation probably accounts for this lack of a decrease. Dissolved Fe, Mn, Cu, Pb, and Zn concentrations were not affected by diversion. The impact of diversion on other metal species is uncertain due to the lack of pre-development data.

Impacts of diversion on water quality in the Burntwood River at the outlet of Threepoint Lake could not be quantified due to the lack of pre-development data for this site.

Burntwood River at Thompson

The only adverse effect of CRD on water quality in the Burntwood River at Thompson was a doubling in turbidity due to bank and shoreline erosion along the Burntwood River below Threepoint Lake. A number of other changes occurred, primarily due to the introduction of softer Churchill River water. These included decreases in conductivity, alkalinity, hardness, reactive Si, dissolved Ca and Mg, dissolved Fe, and extractable Al and Mn. Most changes did not occur until after the autumn of 1977, when the diversion was fully operational, although the initial increase in turbidity and a temporary increase in hardness occurred during CRD construction (June, 1974, to August, 1976), when Rat River flows were cut off at the outlet of Notigi Lake.

Total P concentrations approximately doubled between the construction and immediate post-impoundment periods, probably due to release of P from flooded organic soils and vegetation. Pre-development levels were re-established before 1987 but this does not mean the P release has ended. The annual mean total P concentration in the Burntwood River is still more than twice that in the Churchill River near Leaf Rapids, with most of the P entering the water column as particulates.

Concentrations of dissolved and extractable Al have not been elevated by CRD. A previous report of increased extractable Al following diversion was the product of the mistaken classification of pre-development dissolved Al data as extractable Al.

Split Lake Area

Burntwood River at Inlet to Split Lake

No quantitative assessment of water quality changes in the Burntwood River at the inlet to Split Lake was possible due to the lack of pre-development data.

Nelson River at Inlet to Split Lake

Several significant changes in water quality were noted in the Nelson River above Split Lake between the 1971-1974 pre-development study period and the FEMP study, but none of these changes could be directly attributed to LWR. The FEMP study was conducted during a period of very low Nelson River flows caused by drought conditions in western Canada. The changes could be due, all or in part, to the lower flows.

Nelson River at Outlet of Split Lake

An increase in extractable Fe, such that the annual mean concentration at the outlet now exceeds by 30% the Canadian Water Quality Guidelines for drinking water and protection of aquatic life, has been the only adverse effect of CRD on water quality in Split Lake. This increase, and significant decreases in TDS, DIC, and dissolved As were inferred on the basis of current differences in concentration between the Burntwood and Nelson rivers and the relative contribution of these rivers to total inflow. Statistical analysis of pre- and post-development data for 24 variables revealed significant declines had occurred in 10 (pH, conductivity, hardness (total), alkalinity (total), Ca, Mg, K, Na, Cl, and SO_4). All but the decline in pH were a direct result of diversion.

Churchill River diversion has not caused a significant increase in annual mean water turbidity or suspended sediment concentration (NFR) or a decrease in water transparency despite the much higher suspended sediment concentrations in the Burntwood River than in the Nelson River. Much of the suspended sediment load of the Burntwood River appears to settle out near the mouth, preventing a general increase in suspended sediment levels in the lake or downstream. Sediment deposition also appears to have prevented increased concentrations of extractable Al and total Ni and V in Split Lake.

Water Quality - Water Discharge Relationships

The natural rivers, sites on the Nelson River upstream of Split Lake, and the Burntwood River at Thompson prior to diversion were characterized by having many discharge-correlated variables, with negative correlations dominating. Negatively correlated variables generally included the major cations (Na, K, Ca, Mg) and anions (Cl, F, SO_4) and associated variables (conductivity, alkalinity, hardness, TDS), some metals, DIC, reactive Si, and dissolved N species (TDN, NO_2-NO_3, total NH_3). Positive correlates generally included turbidity, NFR, colour, and the organic carbon species (DOC and TOC).

The discharge relationships were profoundly different at sites affected by CRD. In the Rat River above Threepoint Lake and in the Burntwood River below Threepoint Lake, at Thompson, and at the inlet of Split Lake, all correlations with discharge were positive following diversion. This change is directly attributable to shoreline erosion caused by CRD.

A shift from negative correlations prior to development to positive correlations during the FEMP study also was evident in the Nelson River at the outlet to Split Lake. The role of CRD in this change is unclear due to the unusually low flows on the Nelson River during the period of study.

The water quality - water discharge relationships derived from the FEMP study data offer limited potential for impact prediction or management. The strongest correlations were found at sites where there is the least need for their application; the natural rivers and the Nelson River at Sea River Falls. Most relationships at sites on the CRD were too weak (i.e., $r^2 < 0.25$) to have much predictive ability although there were several important exceptions, including the relationships for NFR and turbidity in the Burntwood River at Thompson and for turbidity and alkalinity in the Burntwood River at the inlet to Split Lake. The weaker discharge relationships at sites on the CRD than at sites off the diversion appear to be due to the relatively less dynamic discharge regime on the CRD.

Mitigation of Adverse Effects

There are few options for mitigation of the adverse impacts of CRD on water quality and considerably more information on the ecological consequences of these impacts is required before any mitigation measure can be recommended. Additional bank stabilization along the Burntwood River might be considered for control of the turbidity problem. Reduction of diversion flows should reduce extractable Fe concentrations in Split Lake, with a 50% reduction in flow required to reach compliance with the current Canadian Water Quality Guidelines.

Recommendations for Further Study

The FEMP study is a major step in documentation of the ecological impacts of the Churchill-Nelson hydroelectric development. This process is by no means complete, however, with a number of outstanding questions remaining. The following activities are recommended to address these questions.

- Investigate the potential for inference of water quality changes in the Jack River and Nelson River below Sea River Falls using pre-development data from upstream sites.

- Examine ecological impacts of elevated turbidity in the Rat-Burntwood system.

- Extend the water quality database for the Burntwood River at Thompson to examine the long-term evolution of water quality on the Churchill River diversion.

- Evaluate the predictive ability of the water quality - water discharge relationships at sites where these relationships appear to have some application.

- Review the applicability of the Canadian Water Quality Guidelines for Fe, Al and Cu to the NFA area and develop site-specific guidelines if necessary.

- Investigate the potential for localized increases in turbidity in Split Lake related to CRD.

- Evaluate the ecological impact of sediment deposition in Split Lake near the mouth of the Burntwood River including the potential for mobilization of associated heavy metals.

- Monitor water quality in Split Lake during periods of higher Nelson River flows for differentiation of water quality changes due to low flows from any changes due to the hydroelectric projects.

Table of Contents

Executive Summary i

Table of Contents xii

List of Tables xvi

List of Figures xxi

List of Appendices xxvii

Acknowledgements xxviii

1.0 Introduction 1
    1.1 Definition of an Adverse Effect 3
    1.2 Limitations to the Assessment of Impacts on Water Quality 5
    1.3 Environment Canada FEMP Study Plan 6
    1.4 Overview of the Churchill-Nelson Hydroelectric Project 8

2.0 Methods 13
    2.1 Physical and Chemical Water Quality 13
        2.1.1 FEMP Sampling Locations 13
        2.1.2 Historical Data Sources and Data Screening 18
    2.2 Statistical Analysis 24
        2.2.1 Water Quality - Water Discharge Relationships 24
        2.2.2 Analysis of Changes in Water Quality . . . 31
        2.2.3 Seasonal Differences 33
    2.3 Bacteriological Water Quality 34
        2.3.1 Data Interpretation 36

3.0 Results and Discussion 37
    3.1 Natural Rivers 37
        3.1.1 Current Water Quality 37
        3.1.2 Variation Among the Rivers 46
        3.1.3 Water Quality - Water Discharge Relationships 46
            3.1.3.1 Burntwood River above Threepoint Lake 52
            3.1.3.2 Footprint River above Footprint Lake 60
        3.1.4 Seasonal Differences 60
            3.1.4.1 Burntwood River above Threepoint Lake 60
            3.1.4.2 Footprint River above Threepoint Lake 68
            3.1.4.3 Aiken River 75
        3.1.5 Long-term Variation in Water Quality . . . 79
        3.1.6 Uses of the Natural River Data 84

3.2 Norway House Area 86
  3.2.1 Nelson River below Sea River Falls 86
    3.2.1.1 Current Water Quality 86
    3.2.1.2 Water Quality - Water Discharge Relationships 86
    3.2.1.3 Seasonal Differences 95
    3.2.1.4 Changes in Water Quality Since Lake Winnipeg Regulation 96
  3.2.2 Jack River 101
    3.2.2.1 Current Water Quality 101
    3.2.2.2 Seasonal Differences 104
    3.2.2.3 Water Quality Changes Since Lake Winnipeg Regulation 104
  3.2.3 Bacterial Studies 107
    3.2.3.1 Norway House 107
    3.2.3.2 Cross Lake 113

3.3 Threepoint Lake and Nelson House Area 118
  3.3.1 Rat River above Threepoint Lake 118
    3.3.1.1 Current Water Quality 118
    3.3.1.2 Water Quality - Water Discharge Relationships 121
    3.3.1.3 Seasonal Differences 125
    3.3.1.4 Changes in Water Quality Since Churchill River Diversion 128
    3.3.1.5 Adverse Effects 135
  3.3.2 Burntwood River at Outlet of Threepoint Lake 135
    3.3.2.1 Current Water Quality 135
    3.3.2.2 Water Quality - Water Discharge Relationships 138
    3.3.2.3 Seasonal Differences 141
  3.3.3 Nelson House Bacterial Testing 141

3.4 Burntwood River at Thompson 147
  3.4.1 Current Water Quality 149
  3.4.2 Water Quality - Water Discharge Relationships 152
    3.4.2.1 Pre-development, 1967-1974 152
    3.4.2.2 Diversion Construction, 1974-1976 . 152
    3.4.2.3 Immediate Post-diversion, 1976-1978 157
    3.4.2.4 Current, 1987-1989 157
  3.4.3 Seasonal Differences 164
  3.4.4 Changes in Water Quality Since Churchill River Diversion 166
  3.4.5 Effect of CRD on Heavy Metals 181
  3.4.6 Adverse Effects 182

3.5 Split Lake Area 184
  3.5.1 Burntwood River at Inlet to Split Lake . . 184
    3.5.1.1 Current Water Quality 184
    3.5.1.2 Water Quality - Water Discharge Relationships 189
    3.5.1.3 Seasonal Differences 193
  3.5.2 Nelson River below Grass River at Inlet to Split Lake 193
    3.5.2.1 Current Water Quality 193
    3.5.2.2 Water Chemistry - Water Discharge Relationships 199
      3.5.2.2.1 Pre-development 199
      3.5.2.2.2 Current 204
    3.5.2.3 Seasonal Differences 207
    3.5.2.4 Changes in Water Quality Since Lake Winnipeg Regulation 211
  3.5.3 Differences in Nelson and Burntwood River Water Quality 214
  3.5.4 Nelson River at Outlet of Split Lake . . . 217
    3.5.4.1 Current Water Quality 217
    3.5.4.2 Water Chemistry - Water Discharge Relationships 220
      3.5.4.2.1 Pre-development 220
      3.5.4.2.2 Current 222
    3.5.4.3 Seasonal Differences 227
    3.5.4.4 Changes in Water Quality Since Lake Winnipeg Regulation and Churchill River Diversion 233
      3.5.4.4.1 Pre/Post Comparison 233
      3.5.4.4.2 Modelled Changes 240
  3.5.5 Adverse Effects 248
  3.5.6 Split Lake Bacterial Testing 251
  3.5.7 York Landing Bacterial Testing 255

4.0 Conclusions 259
  4.1 Current Water Quality 259
  4.2 Temporal Variation in Natural River Water Quality 264
  4.3 Changes in Water Quality Related to the Hydro Developments 265
    4.3.1 Norway House Area 265
    4.3.2 Nelson House and Threepoint Lake Area . . . 265
    4.3.3 Burntwood River at Thompson 268
    4.3.4 Split Lake Area 272
      4.3.4.1 Burntwood River at Inlet to Split Lake 272
      4.3.4.2 Nelson River at Inlet to Split Lake 272
      4.3.4.3 Nelson River at Outlet of Split Lake 274
  4.4 Water Quality - Water Discharge Relationships . . 275
  4.5 Seasonal Variations in Water Quality 277

5.0 Impact Mitigation and Management Options 278
  5.1 Burntwood River - Threepoint Lake to Split Lake . 278
  5.2 Split Lake 279

6.0 Recommendations for Further Study 280
  6.1 Norway House Area 280
  6.2 Burntwood River - Threepoint Lake to Split Lake . 281
  6.3 Split Lake 283
  6.4 Bacteriological Studies 285
  6.5 General Recommendations 286

7.0 References 289

8.0 Glossary of Technical Terms, Chemical Symbols, and Units of Measure 293

List of Tables

1. Description of water quality variables and abbreviations listed in data tables, including the NAQUADAT analysis codes for the FEMP study data. 19

2. Sources of pre-development data used in the analysis of water quality changes. 21

3. Stratification of historical water quality data records by phase of construction. Dates of construction start and completion from G. McCullough, DFO, Winnipeg. 27

4. Sources of discharge data used in the water quality - water discharge analysis. 29

5. Canadian water quality guidelines (CCREM 1987) for drinking water, recreational use, and protection of aquatic life. 38

6. Summary statistics for physical and chemical water quality variables measured in the Burntwood River above Threepoint Lake; May, 1987, to March, 1989. s.d. is standard deviation. 39

7. Summary statistics for physical and chemical water quality variables measured in Footprint River above Footprint Lake; May, 1987, to March, 1989. s.d. is standard deviation. 41

8. Summary statistics for physical and chemical water quality in the Aiken river at the inlet to Split Lake, January, 1987, to October, 1989. s.d. is standard deviation. 43

9. Evaluation of the significance of differences in selected water quality parameters among the Aiken River at the inlet to Split Lake, the Burntwood River above Threepoint Lake and the Footprint River above Footprint Lake. Values listed are the mean and standard deviation (s.d.) for the respective site in 1987 and 1988, and the p value from the Tukey/HSD pairwise comparison. 47

10. Evaluation of the significance of changes in selected water quality parameters in the Burntwood River above Threepoint Lake between 1972-1973 and 1987-1988. X_{pre} denotes the mean for the years 1972 and 1973. X_{post} denotes the mean for the years 1987 and 1988. p is the probability of Student's t-statistic for the comparison. 80

11. Evaluation of the significance of changes in selected water quality parameters in the Footprint River above Footprint Lake between 1972-1973 and 1987-1988. X_{pre} denotes the mean for the years 1972 and 1973. X_{post} denotes the mean for the years 1987 and 1988. p is the probability of Student's t-statistic for the comparison. 81

12. Summary statistics for physical and chemical water quality variables measured in the Nelson River below Sea River Falls; January, 1987, to September, 1989. s.d. is standard deviation. 87

13. Summary statistics for physical and chemical water quality variables measured in the Nelson River below Sea River Falls; June, 1971, to May, 1972. s.d. is standard deviation. 100

14. Summary statistics for physical and chemical water quality variables measured in the Jack River (West Channel); January, 1987, to September, 1989. s.d. is standard deviation. 102

15. Summary statistics for physical and chemical water quality variables measured in the Rat River above Threepoint Lake; September, 1986, to October, 1989. s.d. is standard deviation. 119

16. Evaluation of the significance of changes in selected water quality parameters in the Rat River above Threepoint Lake following Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1972 and 1973. X_{post} denotes the mean for post-development years 1987 through 1989. p is the probability of Student's t-statistic for the comparison. 129

17. Comparison of water quality in the Rat River before (1972-1973) and after (1987-1989) Churchill River diversion with water quality in the Churchill River near Leaf Rapids (1972-1973). All values are annual means. Units are mg L^{-1} except where noted. 130

18. Summary statistics for physical and chemical water quality variables measured in the Burntwood River at the outlet to Threepoint Lake; September, 1986, to October, 1989. s.d. is standard deviation. 136

19. Summary statistics for physical and chemical water quality in the Burntwood River at Thompson, January, 1987, to October, 1989. s.d. is standard deviation. 150

20. Evaluation of the significance of changes in selected water quality parameters in the Burntwood River at Thompson following Churchill River diversion. Values listed are the mean and standard deviation (s.d.) for the respective period and the p value from the Tukey/HSD pairwise comparison. Pre denotes the pre-development period from 1967 to May 1974. Const denotes the construction period between June 1974 and August 1976. Post is the immediate post-diversion period from September 1976 to December 1977. Current represents the FEMP study period between January 1987 and October 1989. 167

21. Comparison of current water quality in the Burntwood River at Thompson and at the inlet to Split Lake. X_{thm} denotes the 1987-1989 mean for the Burntwood River at Thompson. X_{spl} denotes the 1987-1989 mean for the Burntwood River at the inlet to Split Lake. p is the probability of Student's t-statistic for the comparison. 185

22. Summary statistics for physical and chemical water quality in the Burntwood River at the inlet to Split Lake, September, 1986, to October, 1989. s.d. is standard deviation. 187

23. Comparison of current water quality in the Nelson River below Sea River Falls and at the inlet to Split Lake. X_{srf} denotes the 1987-1989 mean for the Nelson River below Sea River Falls. X_{spl} denotes the 1987-1989 mean for the Nelson River at the inlet to Split Lake. p is the probability of Student's t-statistic for the comparison. 195

24. Summary statistics for physical and chemical water quality in Nelson River below Grass River, September, 1986, to October, 1989. s.d. is standard deviation. 197

25. Evaluation of the significance of changes in selected water quality parameters in the Nelson River near the inlet of Split Lake following Lake Winnipeg regulation. X_{pre} denotes the mean for pre-development years 1972 and 1973. X_{post} denotes the mean for the post-development years 1987 through 1989. p is the probability of Student's t-statistic for the comparison. 212

26. Comparison of current water quality in the Nelson and Burntwood rivers at the inlet to Split Lake. X_{nel} denotes the 1987-1989 mean for the Nelson River. X_{bwr} denotes the 1987-1989 mean for the Burntwood River. p is the probability of Student's t-statistic for the comparison. 215

27. Summary statistics for the physical and chemical water quality in the Nelson River at the outlet of Split Lake, September, 1986, to October, 1989. s.d. is standard deviation. 218

28. Evaluation of the significance of changes in selected water quality parameters in the Nelson River near the outlet of Split Lake following Churchill River diversion and Lake Winnipeg regulation. X_{pre} denotes the mean for the pre-development years 1972 and 1973. X_{post} denotes the mean for the post-development years 1987 through 1989. p is the probability of Student's t-statistic for the comparison. 234

29. Comparison of observed and predicted values of selected water quality parameters in the Nelson River at the outlet of Split Lake following Churchill River diversion. Pre- and post-development predicted values have been calculated as in Figure 44. BWR denotes the Burntwood River. NR denotes the Nelson River. All values in mg L^{-1}, except where noted. 244

30. Comparison of current water quality in the Nelson River at the inlet and outlet to Split Lake. X_{in} denotes the 1987-1989 mean for the Nelson River at the inlet to Split Lake. X_{out} denotes the 1987-1989 mean for the Nelson River at the outlet to Split Lake. p is the probability of Student's t-statistic for the comparison. 245

31. Water quality variables in the natural rivers which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period. 260

32. Water quality variables at sites in the Norway House area which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period. 261

33. Water quality variables at sites on the Churchill River Diversion which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period. 262

34. Water quality variables at sites in the Split Lake area which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period. 263

35. Water quality variables which changed significantly following Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1971 through 1974. X_{post} denotes the mean for the post-development years 1987 through 1989. 266

36. Water quality variables in the Burntwood River at Thompson which changed significantly following Churchill River diversion. X_{pre} denotes the pre-development period from 1967 to May 1974. X_{const} denotes the construction period between June 1974 and August 1976. X_{post} is the immediate post-diversion period from September 1976 to December 1977. $X_{current}$ represents the FEMP study period between January 1987 and October 1989. 269

37. Water quality changes at sites in the Split Lake area following Lake Winnipeg regulation and Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1972 and 1973. X_{post} denotes the mean for the post-development years 1987 through 1989. For modelled changes, X_{pre} is the current concentration in the Nelson River at the inlet to Split Lake. 273

List of Figures

1. Churchill and Nelson rivers hydroelectric development, indicating the altered flow regime of the rivers. Dark tone indicates relative magnitude of lower Churchill discharge remaining after diversion; mid-tone indicates portion of Churchill River discharge diverted at Southern Indian Lake; light tone indicates Nelson River discharge. From Newbury et al. (1984). 7

2. Historical annual mean discharge in the Nelson River at Kelsey Generating Station and in the Burntwood River at Thompson, 1961-1989. Data from Water Survey of Canada (1990). 11

3. Location of FEMP water quality study areas. See Figures 4, 5, and 6 for location of FEMP and pre-development sampling locations. 14

4. Location of FEMP and pre-development water quality sampling stations in the Nelson House study area. 15

5. Location of FEMP and pre-development water quality sampling stations in the Split Lake study area. 16

6. Location of the FEMP and pre-development water quality sampling stations in the Norway House study area. 17

7. Graphical representation of the procedure used to standardize chemistry data to discharge. See text for further definition of terms. Figure modified from Jones et al. (1985). 30

8. Water chemistry - water discharge relationships in the Burntwood River above Threepoint Lake, 1987-1989.
a. field conductivity b. lab conductivity c. field turbidity d. colour e. TDS f. total hardness
g. total NFR h. fixed NFR i. total alkalinity
j. bicarbonate alkalinity k. TDN l. dissolved NO_2-NO_3
m. total NH_3 n. DOC o. TOC p. DIC q. dissolved Ca
r. dissolved Mg s. dissolved K t. dissolved Na
u. reactive Si v. dissolved Cl w. dissolved Fe
x. extractable Mn y. dissolved Mn. 53

9. Water chemistry - water discharge relationships in the Footprint River above Footprint Lake, 1987-1989.
a. field conductivity b. lab conductivity c. colour
d. TDS e. total NFR f. fixed NFR g. total alkalinity
h. bicarbonate alkalinity i. total hardness j. TDN
k. total N l. DIC m. dissolved Mg n. dissolved Na
o. dissolved F p. dissolved As q. dissolved Ni. 61

10. Seasonal water quality trends in the Burntwood River above Threepoint Lake. a. colour b. chlorophyll a c. dissolved Na d. dissolved Mg e. dissolved Zn. 66

11. Seasonal water quality trends in the Footprint River above Footprint Lake. a. lab conductivity b. DIC c. chlorophyll a d. total alkalinity e. bicarbonate alkalinity f. total hardness g. dissolved Na h. reactive Si i. dissolved Cl j. dissolved Ca k. dissolved Mg l. dissolved K m. dissolved Cu n. dissolved Zn o. dissolved Fe p. extractable Mn q. dissolved Mn. 69

12. Seasonal water quality trends in the Aiken River. a. chlorophyll a b. dissolved As c. dissolved Zn d. dissolved Ca e. total NH_3 f. DIC g. total alkalinity h. bicarbonate alkalinity i. total hardness. 76

13. Water chemistry - water discharge relationships in the Nelson River below Sea River Falls, 1987-1989. a. dissolved Cl b. dissolved SO_4 c. extractable Fe d. dissolved Ca e. dissolved Mg f. dissolved K g. dissolved Na h. dissolved NO_2-NO_3 i. total NH_3 j. POC k. DIC l. bicarbonate alkalinity m. particulate N n. total hardness o. non-carbonate hardness p. TDS q. total alkalinity r. total NFR s. fixed NFR t. lab pH u. field turbidity v. field conductivity w. lab conductivity. 89

14. Seasonal water quality trends in the Nelson River below Sea River Falls. a. total NFR b. fixed NFR c. dissolved Mg d. reactive Si. 97

15. Seasonal water quality trends in the Jack River (West Channel) near Norway House. a. total NFR b. chlorophyll a c. dissolved Cd d. dissolved Ni. 105

16. Fecal coliform bacteria counts from the seasonal study in the vicinity of Norway House. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. See Appendix 1 for site locations and descriptions. a. Asiniwapiska b. Bridge into Fort Island c. Government Dock d. Green Dock e. Historic Dock at Bay Store Bridge f. School Dock g. Sewage Outlet Bay at Church Point. 108

17. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Norway House Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 11 August 1987 b. Sampled 24 August 1987. See Appendix 1 for site descriptions and locations. 112

18. Fecal coliform bacteria counts from the seasonal study in the vicinity of Cross Lake. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Beardy's Point b. Saggitawak Bridge c. Saggitawak Swimming Area d. Wapak Swimming Area e. Left of the Water Treatment Plant Intake. See Appendix 1 for site descriptions and locations. 114

19. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Cross Lake Water Treatment Plant Intake. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 11 August 1987 b. Sampled 25 August 1987. See Appendix 1 for site descriptions and locations. 117

20. Water chemistry - water discharge relationships in the Rat River above Threepoint Lake, 1987-1989. a. total alkalinity b. bicarbonate alkalinity c. TDS d. total hardness e. ortho PO_4 f. particulate N g. POC h. dissolved Ca i. dissolved Se. 122

21. Seasonal water quality trends in the Rat River above Threepoint Lake. a. dissolved NO_2-NO_3 b. TDN c. total N d. chlorophyll a e. dissolved Zn f. total Ni. 126

22. Water chemistry - water discharge relationships in the Burntwood River at the outlet of Threepoint Lake, 1987-1989. a. lab conductivity b. TDS c. total hardness d. total alkalinity e. bicarbonate alkalinity f. extractable Al. 139

23. Seasonal water quality trends in the Burntwood River at the outlet of Threepoint Lake. a. dissolved Zn b. dissolved NO_2-NO_3 c. TDN. 142

24. Fecal coliform bacteria counts from the seasonal study in the vicinity of Nelson House. Mean is the geometric mean. Dotted line indicates Canadian Water Quality Guideline. a. Church Bay (East Ramp) b. Government Dock c. Industrial Bay d. Inspiration Point e. Metis Beach f. New Development Area (Across from Cemetery) g. Nursing Station Bay h. School Bay. 143

25. Fecal coliform bacteria counts from the intensive study in the vicinity of the Nelson House Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 12 August 1987 b. Sampled 25 August 1987. 148

26. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the pre-development period; from February 1967 to May 1974. a. total alkalinity b. total hardness c. lab conductivity d. lab turbidity e. dissolved Na f. reactive Si. 153

27. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the construction period; from June 1974 to August 1976. a. lab conductivity b. total alkalinity c. total hardness d. dissolved Mg e. dissolved Na. 155

28. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the immediate post-diversion period; from September 1976 to December 1977. a. pH b. total alkalinity c. total hardness d. reactive Si. 158

29. Water chemistry - water discharge relationships in the Burntwood River at Thompson, 1987-1989. a. total NFR b. fixed NFR c. lab turbidity d. total alkalinity e. bicarbonate alkalinity f. particulate P g. total P h. dissolved K i. dissolved Na j. total Cu k. extractable Fe l. extractable Mn m. dissolved Se n. total Zn. 159

30. Seasonal trends in chlorophyll a concentrations in the Burntwood River at Thompson. 165

31. Temporal trends in (a) NFR, (b) turbidity, (c) alkalinity, (d) hardness, and (e) total P in the Burntwood River at Thompson based on data analyzed by Playle and Williamson (1986) and in this study. 175

32. Water chemistry - water discharge relationships in the Burntwood River at the inlet to Split Lake, 1987-1989. a. field turbidity b. lab turbidity c. total alkalinity d. bicarbonate alkalinity e. free CO_2 f. DOC g. TOC h. dissolved Na i. dissolved Mn j. dissolved Se k. dissolved Zn. 190

33. Seasonal water quality trends in the Burntwood River at the inlet to Split Lake. a. dissolved NO_2-NO_3 b. chlorophyll a. 194

34. Water chemistry - water discharge relationships in the Nelson River below Sipiwesk Lake, 1972-1973. a. NFR. 200

35. Water chemistry - water discharge relationships in the Nelson River above Kelsey Generating Station, 1972-1974. a. field conductivity b. TDS c. POC d. dissolved Na e. dissolved SO_4 f. dissolved Fe g. dissolved Ca h. dissolved Mg i. dissolved K. 201

36. Water chemistry - water discharge relationships in the Nelson River below Kelsey Generating Station, 1972-1974. a. field conductivity b. TDS c. NFR d. TDN. 203

37. Water chemistry - water discharge relationships in the Nelson River at the inlet to Split Lake, 1987-1989. a. ortho PO_4 b. total N c. dissolved F d. dissolved SO_4 e. dissolved Cd f. total Cd. 205

38. Seasonal water quality trends in the Nelson River at the inlet to Split Lake. a. lab conductivity b. dissolved Mg c. dissolved K d. total hardness e. dissolved NO_2-NO_3 f. DIC. 208

39. Water chemistry - water discharge relationships in Split Lake at Split Lake, 1972-1974. a. lab conductivity b. total alkalinity c. total hardness d. dissolved K. 221

40. Water chemistry - water discharge relationships near the outlet of Split Lake, 1972-1973. a. field conductivity b. dissolved Na c. dissolved K d. dissolved Mg e. dissolved SO_4 f. TDS. 223

41. Water chemistry - water discharge relationships in the Nelson River at the outlet of Split Lake, 1987-1989. a. DOC b. TOC c. dissolved Fe. 225

42. Seasonal water quality trends in the Nelson River at the outlet of Split Lake. a. dissolved NO_2-NO_3 b. TDN c. total N d. colour e. dissolved SO_4 f. dissolved Zn g. total hardness h. lab conductivity i. total alkalinity j. dissolved Ca k. dissolved K l. dissolved Na. 228

43. Temporal trends in (a) dissolved Na, (b) turbidity, and (c) dissolved Cl in the Nelson River at the outlet to Split Lake based on data analyzed by Playle and Williamson (1986) and the data analyzed in this study. 238

44. (A). Simplified model of Split Lake hydrology, and (B) derivation of numerical models used to predict the concentration of a substance in the Nelson River near the outflow of Split Lake based on the concentrations in the inflowing Burntwood and Nelson rivers and the relative contributions of these rivers to discharge at the outflow. Model predictions are listed in Table 29. 242

45. Fecal coliform bacteria counts from the seasonal study in the vicinity of Split Lake. Mean is the geometric mean. Dotted line indicates Canadian Water Quality Guideline. a. Bay Store Beach Area b. Beach Point between Nursing Station and Bay Store c. First Beach d. Nursing Station Bay e. Second Beach near Ferry. See Appendix 1 for site locations and descriptions. 252

46. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Split Lake Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 12 August 1987 b. Sampled 26 August 1987. See Appendix 1 for site locations and descriptions. 256

47. Fecal coliform bacteria counts from the seasonal study in the vicinity of York Landing. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Ferry Dock b. Gravel Pit c. Island Beach d. Water Treatment Plant. See Appendix 1 for site locations and descriptions. 257

List of Appendices

1. Bacteriological study methods. 297

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1.0 Introduction

Environment Canada and the Department of Fisheries and Oceans undertook in 1986 a five year program of ecological monitoring in northern Manitoba which has come to be known as the Federal Ecological Monitoring Program (FEMP). The overall purpose of this program is to address Canada's obligations under the Northern Flood Agreement (NFA) with respect to requirements for ecological monitoring and the identification and disclosure of adverse effects of Churchill River diversion (CRD) and Lake Winnipeg regulation (LWR).

The NFA was signed between Canada, Manitoba, Manitoba Hydro, and the Northern Flood Committee (NFC) which represents the 5 Indian Bands affected by the Lake Winnipeg Regulation and the Churchill-Nelson Rivers Diversion Project (LWCNR). The objective of the NFA is to ensure that all persons represented by the NFC who are adversely affected by the project receive fair and equitable treatment.

This report presents the results of the water quality monitoring program implemented by Environment Canada as part of the FEMP. Existing water quality is described, changes in physical and chemical water quality since the LWCNR project began operation are documented and, where possible, the cause of these changes is determined. Any adverse effects directly resulting from changes in water quality caused by the Project are described.

The specific objectives of the study were to:

1.) document and assess changes in water quality which have or are continuing to occur since the hydroelectric developments in the study area were completed;

2.) document and assess changes in water quality resulting from the developments which have had an adverse effect on the water uses of the five NFA communities;

3.) identify water quality conditions which exceed generally accepted national guidelines for identified water uses, but for which there is insufficient information to identify cause, and;

4.) develop and recommend a water quality management program which will provide information necessary to document future changes in water quality and for use by water quality managers.

The purpose of these objectives is to fulfil the water quality aspect of Canada's obligations under Clauses 17.5 and 17.1 of the NFA. Clause 17.5 refers to monitoring adverse effects of the project and states "In particular but without limitation, monitoring of adverse effects of the Project pursuant to the Lake Winnipeg, Churchill and Nelson Rivers Study Board recommendations shall be planned and implemented so as to provide such information as may be necessary to give effect to this agreement". Clause 17.1 states, "Hydro and Canada and Manitoba, severally and jointly, undertake to implement such recommendations of the Lake Winnipeg, Churchill, and Nelson Rivers Study Board Report which affect the Communities and which fall within their respective or joint jurisdictions". One such recommendation is recommendation Number 10 which states "That appropriate government departments and agencies develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the affected area".

Thus, the two primary tasks of the water quality monitoring program were to document existing water quality, as required by Clause 17.1 (recommendation 10), and to identify adverse effects, as required by Clause 17.5.

1.1 Definition of an Adverse Effect

The definition of an adverse effect employed in this study is a change in water quality which has a detrimental effect on one or more of the traditional water uses of the five NFA communities (Seidl 1987). Traditional water uses of the communities which are considered in this report include drinking water, recreation, and aquatic life. Aesthetic impacts also were considered. As suggested by Seidl (1987), the definition of a detrimental effect employed in this report is a significant change in water quality since LWCNR Project construction, which is not attributable to natural causes or to causes other than the hydroelectric project, and which causes one or more parameters to exceed the Canadian Water Quality Guidelines (CCREM 1987) for drinking water, recreational water use, and protection of aquatic life.

The CCREM guidelines have been established by the Canadian Council of Resource and Environment Ministers to provide basic information about the effects of water quality parameters on uses in order to assess water quality issues and concerns and to establish water quality objectives for specific sites (CCREM 1987). The guidelines were developed following review of existing guidelines from many sources. Those guidelines considered to be appropriate for Canadian environmental conditions were adopted. Those found inappropriate, were either adopted after modification on the basis of scientific information specific to Canadian conditions or, if no data existed on which to base a modification, no guideline was established.

Use of the CCREM guidelines to define detrimental or harmful changes in water quality and, hence, adverse effects represents a balance between the need to prevent harm to biota or water users, while recognizing that most systems have some capacity to absorb or assimilate some change without causing any harm. The CCREM guideline level for a variable is set below the level at which physiological effects have been shown to first appear in the most sensitive species and life stage which have been tested. The guidelines therefore protect all forms of aquatic life and all life stages during indefinite exposure (Williamson 1988).

Use of the CCREM guidelines to define an adverse effect is somewhat limiting in that guidelines have not been established for all variables, the guidelines are not based on site-specific cause-effect studies in northern Manitoba, and they do not take into account the effect of cumulative stresses caused by the simultaneous exceedence of guideline values by two or more water quality variables. Modification of the guidelines to accommodate cumulative effects was beyond the scope of this study. Site-specific information on biological responses to the water quality changes which have occurred at the study sites is largely non-existent, necessitating reliance on the more general guidelines. The documentation of adverse effects of hydroelectric development in northern Manitoba should be considered as an ongoing process; information on the nature of changes in one environmental component (e.g., water quality) often is necessary to focus subsequent studies in other areas (e.g., benthos or fish feeding efficiency). Future studies may reveal adverse effects which are not presently apparent.

1.2 Limitations to the Assessment of Impacts on Water Quality

The scope and findings of any retrospective analysis of changes in water quality, such as that presented here, are dependent upon the nature and extent of pre-development data. While it may be possible in some cases to estimate pre-development concentrations for some variables, most statements of significant change are dependent upon the existence of sufficient pre-development data, of comparable quality, for the application of a statistical test. Unfortunately, the pre-development water quality database is not complete for any of the sites considered in this report. In particular, the pre-development nutrient and metals databases were incomplete for most sites. Specific data limitations are identified throughout the text.

1.3 Environment Canada FEMP Study Plan

A water quality monitoring program was implemented in late 1986 to document existing water quality. This program was developed in co-operation with the Department of Fisheries and Oceans, Manitoba Natural Resources, and Manitoba Environment to provide the most complete coverage of the NFA area that was possible within the overall program budget. Results of studies by Manitoba Environment are presented in Playle (1986), Playle and Williamson (1986), Duncan and Williamson (1988), and Ralley and Williamson (1990). Data gathered by Manitoba Natural Resources are documented in Green (1990) and are summarized and analyzed by Ramsey et al. (1989) and Berger and Ramsey (1991).

The Environment Canada program included a three year study of physical and chemical water quality at 11 sites on the Rat, Burntwood, Footprint, upper and lower Nelson, Jack, and Aiken rivers to document current conditions (Figure 1). All data collected under this program are documented in Strange (1991). In this report, the FEMP study data are compared with all available pre-development data for the sites to identify statistically

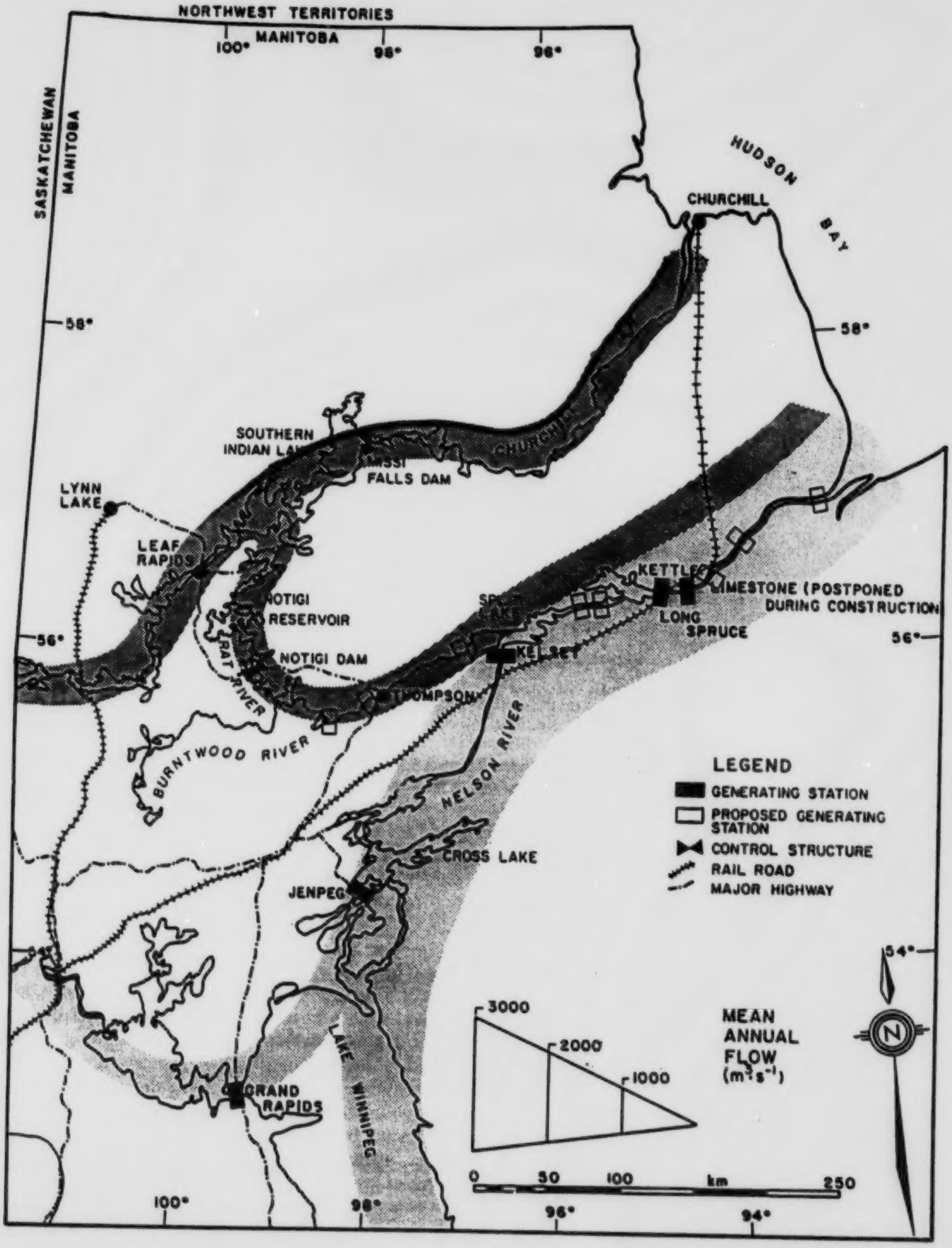

NORTHWEST TERRITORIES
MANITOBA
100°
98°
96°
SASKATCHEWAN
MANITOBA
HUDSON BAY
CHURCHILL
58°
58°
SOUTHERN INDIAN LA
MISSI FALLS DAM
CHURCHILL
LYNN LAKE
LEAF RAPIDS
NOTIGI RESERVOIR
NOTIGI DAM
RAT RIVER
THOMPSON
KELSEY
SPLIT LAKE
KETTLE
LIMESTONE (POSTPONED DURING CONSTRUCTION)
LONG SPRUCE
56°
56°
BURNTWOOD RIVER
NELSON RIVER
CROSS LAKE
JENPEG
GRAND RAPIDS
LAKE WINNIPEG
LEGEND
■ GENERATING STATION
☐ PROPOSED GENERATING STATION
▶◀ CONTROL STRUCTURE
RAIL ROAD
MAJOR HIGHWAY
54°
3000
2000
1000
MEAN ANNUAL FLOW (m^3s^{-1})
0
50
100
km
250
100°
98°
96°
94°
N

Figure 1. Churchill and Nelson rivers hydroelectric development, indicating the altered flow regime of the rivers. Dark tone indicates relative magnitude of lower Churchill discharge remaining after diversion; mid-tone indicates portion of Churchill River discharge diverted at Southern Indian Lake; light tone indicates Nelson River discharge. From Newbury et al. (1984).

significant changes.

In addition to the study of current physical and chemical water quality, Environment Canada undertook a study of the bacteriological quality of recreational waters in the 5 NFA communities. The bacterial studies were intended specifically for evaluation of the present quality of waters for recreational (principally swimming) uses. The studies were not designed nor intended for evaluation of the effects of the LWCNR Project on bacterial water quality.

1.4 Overview of the Churchill-Nelson Hydroelectric Project

The Churchill and Nelson rivers drain 1.4 million km^2 of the interior plains of western North America, flowing northeasterly across the Canadian Shield to Hudson Bay. In the mid-1960's, joint Federal-Provincial studies of the feasibility of developing hydroelectric generating capacity on the Churchill and Nelson rivers for markets in southern Canada and the northern United States identified 9 dam sites on the Nelson River (Figure 1). No development of generating stations on the Churchill River was recommended. Instead, diversion of a significant portion of Churchill River flows was proposed to increase flows through the Nelson River dams (Newbury et al. 1984).

A number of diversion schemes were considered but the plan which was implemented in 1976 involved a 3 m rise in the mean level of Southern Indian Lake, diverting 75% (about 835 m^3 sec^{-1}) of Churchill River flow to the Nelson River via the Rat and Burntwood

Rivers (Figure 1). Churchill River diversion required the construction of a control dam at Missi Falls, the natural outlet of SIL, the excavation of a short diversion canal between South Bay on SIL and Issett Lake, and the construction of a control structure at the outlet of Notigi Lake which was used to create and now regulates the Notigi Reservoir.

Construction and operation of the diversion have caused dramatic changes in the hydrologic regime of the Rat River. River flow was completely cut off at the Notigi Dam site from June 1974 to August 1976 to facilitate construction of the dam and filling of the Notigi Reservoir (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.). The reservoir incorporates several major lakes, flooding in excess of 500 km^2 of forested land in the Rat River valley. With the beginning of diversion operation in September, 1976, annual mean flows below Notigi Dam increased from approximately 30 m^3 sec^{-1} (Water Resources Branch 1974) to about 865 m^3 sec^{-1} (Manitoba Hydro, unpubl. data).

The Rat River merges with the Burntwood River at Threepoint Lake (Figure 1). Prior to diversion construction, the Rat River contributed approximately 30% of total Burntwood River flows below Threepoint Lake (Water Resources Branch 1974). This contribution accounts for the approximately 30% reduction in Burntwood River flow at Thompson during Notigi Dam construction (Water Survey of Canada 1990).

Diversion produced a substantial increase in mean Burntwood River flows below Threepoint Lake, from an average of about 120 m^3

sec^{-1} between 1958 and 1973, to about 900 m^3 sec^{-1} between 1978 and 1989 (Figure 2). The increase in flow due to diversion developed gradually over a period of about 1 year, reaching full flows in September, 1977.

The increased flows on the lower Rat and Burntwood rivers following diversion produced elevated water levels on the rivers and lakes causing flooding along shorelines, backflooding along stream channels, and incorporation of, and marginal flooding along, several adjacent lakes. Mean levels on Threepoint and Footprint lakes increased approximately 4.5 m. Approximately 99 km^2 of land was flooded between the Notigi Dam and the inlet to Split Lake (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.).

Churchill River diversion caused an approximately 30% increase in mean Nelson River flows below Split Lake, although the relative contribution of the CRD to Nelson River flows varies greatly from year to year. Prior to diversion, mean annual Burntwood River flow at the inlet to Split Lake was approximately 5% of mean Nelson River inflow to the lake (Figure 2). Since diversion in 1976, mean Burntwood River flow has ranged from 29% to 100% of Nelson River flow, with this broad range in relative contribution due primarily to discharge fluctuations on the Nelson River. During the 1987-1989 period of the FEMP study, Burntwood River flows relative to Nelson River flows were the highest since the diversion began operating in 1976. The impact of CRD on water chemistry in Split Lake and sites downstream is dependent on the relative contribution of CRD to

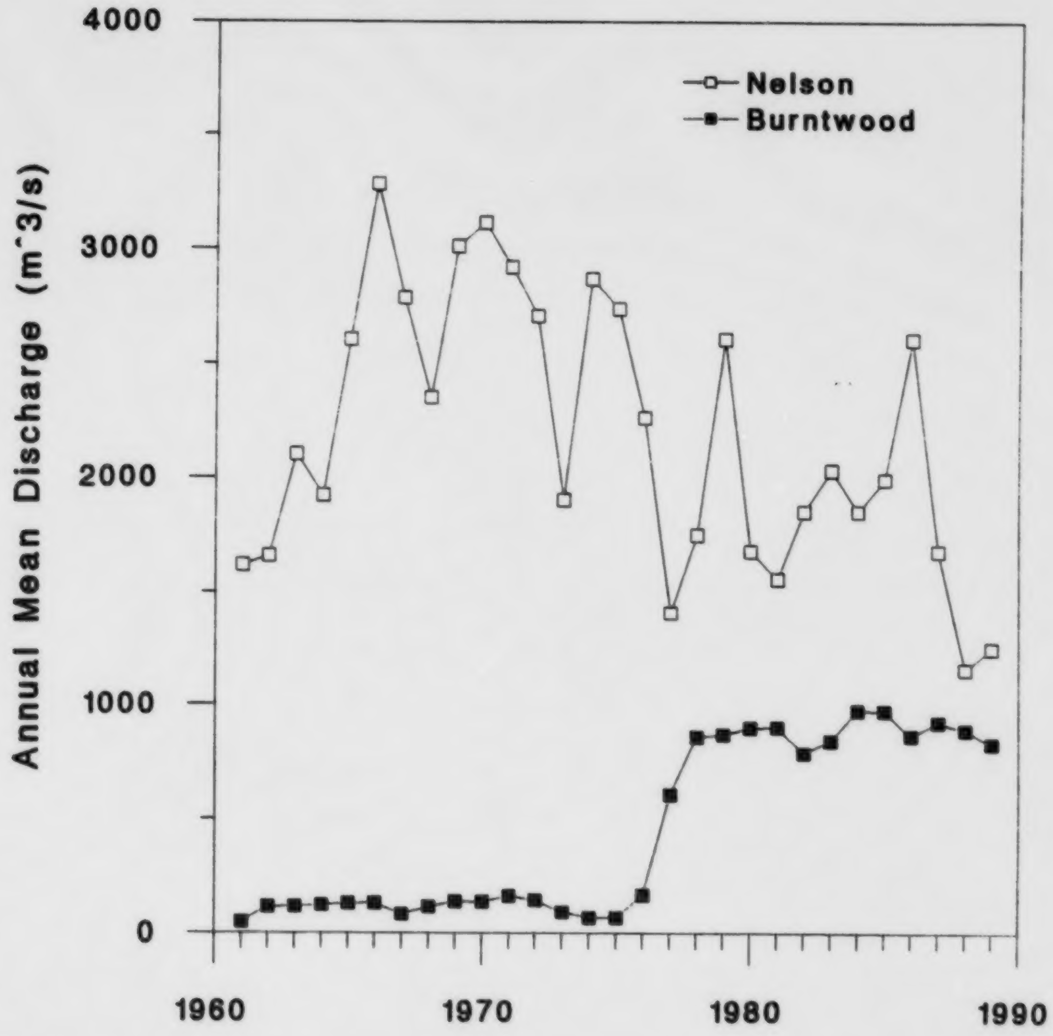


Figure 2. Historical annual mean discharge in the Nelson River at Kelsey Generating Station and in the Burntwood River at Thompson, 1961-1989. Data from Water Survey of Canada (1990).

Nelson River flows (Section 3.5.4). Thus, the changes in Split Lake water quality which are documented in this report probably are more severe than any which occurred in the first 10 years following diversion.

Another significant component of the Churchill-Nelson project is the regulation of levels on 24,400 km^2 Lake Winnipeg by the Jenpeg Dam, located on the west channel of the Nelson River above Cross Lake, for dependable mid-winter flows (Figure 1; Newbury et al. 1984). The Lake Winnipeg regulation component of the project also involved alteration of the water flow pattern in southern Playgreen Lake, due to the construction of Two Mile Channel, and in northern Playgreen Lake by Eight Mile Channel, which now diverts water through Kiskittogisu Lake. The dam maintains water levels on Lake Winnipeg and Playgreen and Kiskittogisu lakes near the high end of the natural range by increasing the mean level and decreasing the range of fluctuation. Backflooding is restricted to lower Kiskittogisu Lake and the downstream reach above the dam. Little Playgreen Lake and the East Channel of the Nelson River are probably subject to slightly higher than natural flows and levels due to increased average Playgreen Lake levels at the outlets into Little Playgreen Lake (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.).

2.0 Methods

2.1 Physical and Chemical Water Quality

2.1.1 FEMP Sampling Locations

The physical and chemical water quality sampling program concentrated on three major areas; the Nelson House area on the CRD between Threepoint Lake and Thompson, the major inlets and outlets of Split Lake, and the Norway House area (Figure 3).

In the Threepoint Lake area (Figure 4), stations were situated on the Burntwood River, above the confluence with the Wimapedi River, on the Rat River immediately above Threepoint Lake, on the Footprint River above Footprint Lake, at the outlet of Threepoint Lake, and on the Burntwood River at Thompson, approximately mid-way between Threepoint and Split lakes. In the Split Lake area (Figure 5), stations were situated on the Nelson River above Split Lake, about 1 km below the confluence with the Grass River; on the Burntwood River above Split Lake, about 1 km below the confluence with the Odei River; at the mouth of the Aiken River; and, at the outlet of Split Lake. In the Norway House area (Figure 6), samples were collected from the Jack River at Norway House and from the Nelson River below Sea River Falls.

The sites on the Burntwood River above Threepoint Lake, the Footprint River, and the Aiken River have not been affected by the hydroelectric developments and were included to characterize and provide a measure of the natural variation in water quality on these smaller regional rivers.

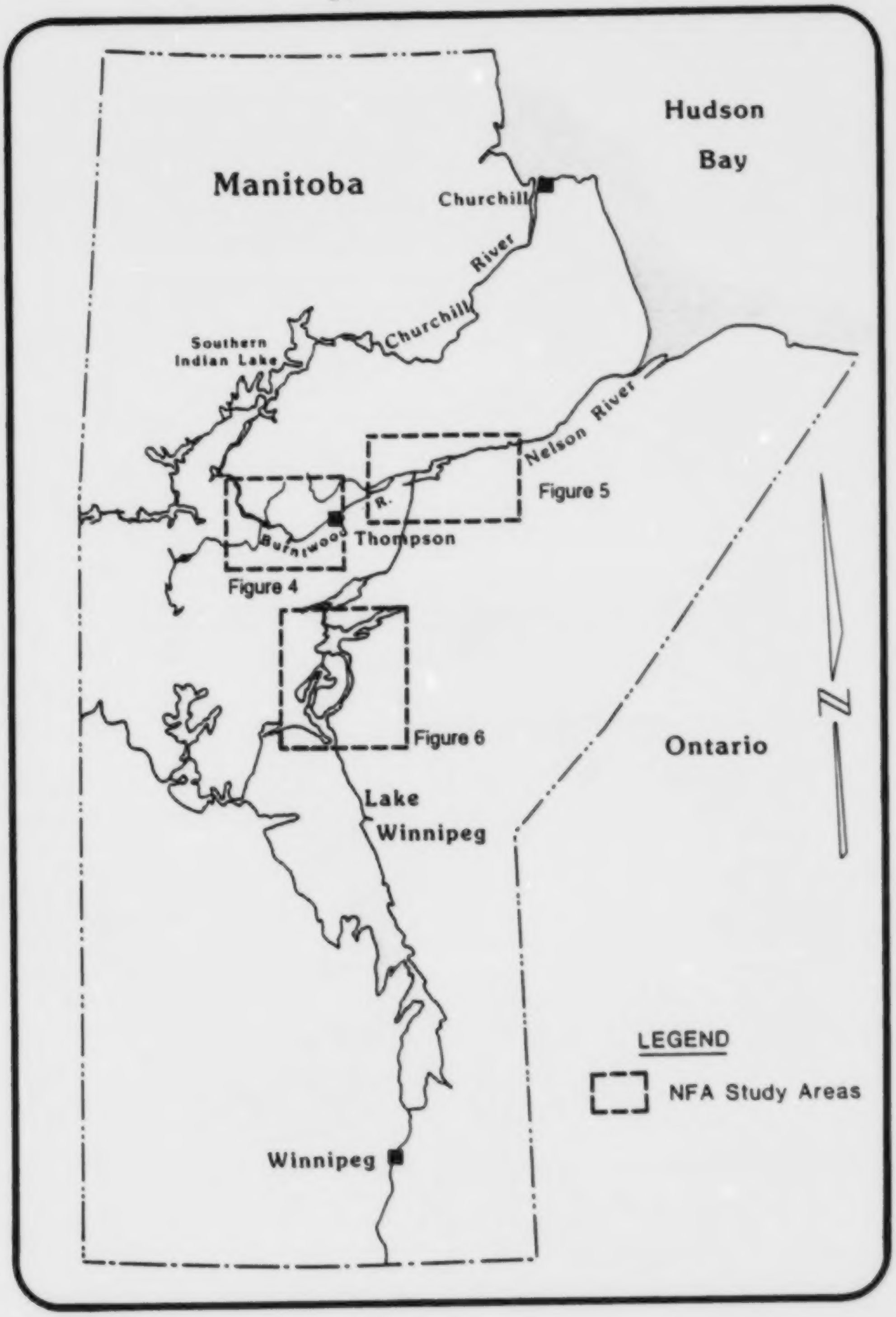

Hudson Bay
Manitoba
Churchill
Churchill River
Southern Indian Lake
Nelson River
Figure 5
Burntwood R.
Thompson
Figure 4
Figure 6
Ontario
N
Lake Winnipeg
LEGEND
NFA Study Areas
Winnipeg

Figure 3. Location of FEMP water quality study areas. See Figures 4, 5, and 6 for location of FEMP and pre-development sampling locations.

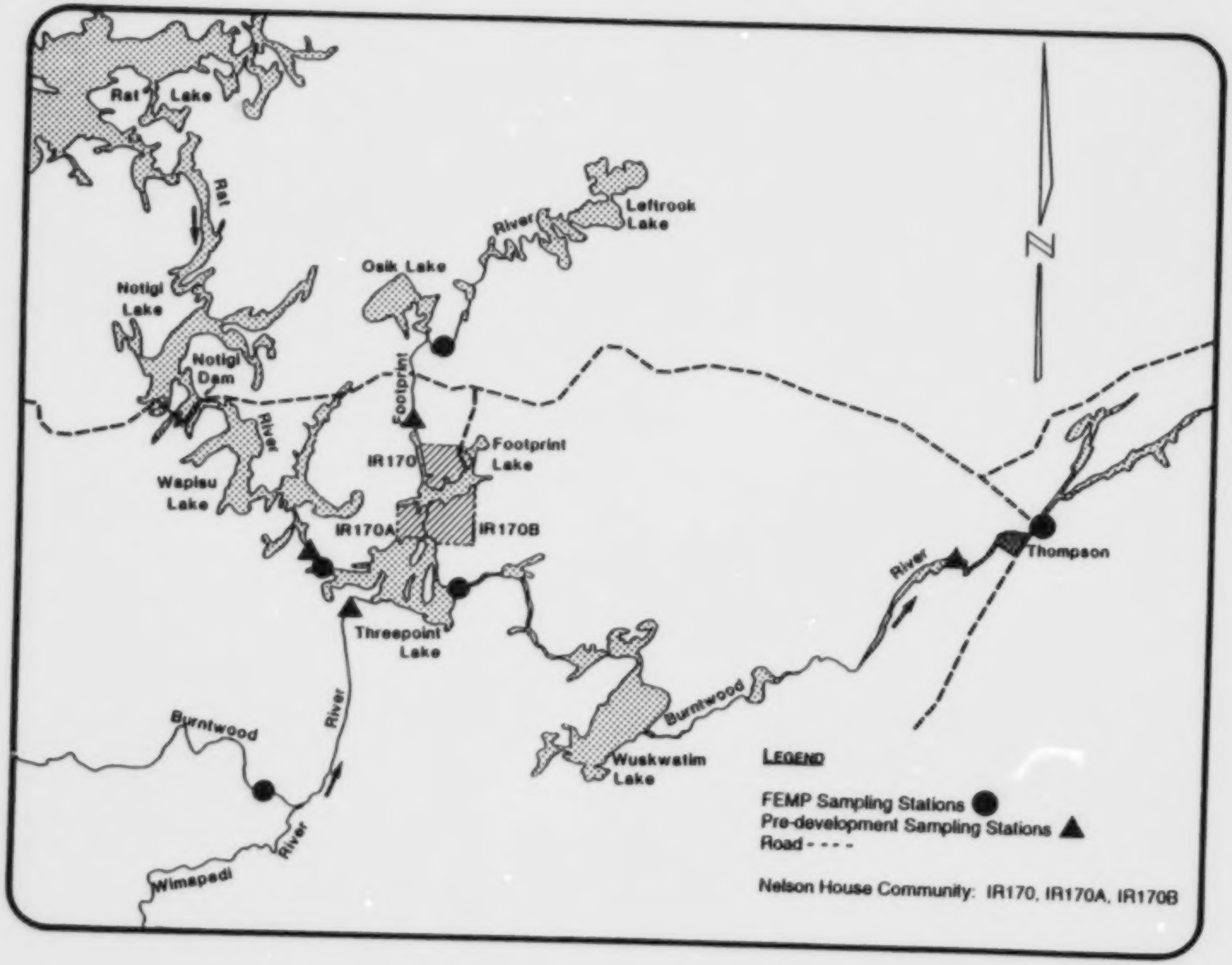

N
Rat Lake
Rat
Notigi Lake
Notigi Dam
River
Wapisu Lake
Osik Lake
River
Leftrook Lake
Footprint
Footprint Lake
IR170
IR170A
IR170B
Threepoint Lake
River
Burntwood
River
Wimapedi
Burntwood
Wuskwatim Lake
River
Thompson
LEGEND
FEMP Sampling Stations ●
Pre-development Sampling Stations ▲
Road - - - -
Nelson House Community: IR170, IR170A, IR170B

Figure 4. Location of FEMP and pre-development water quality sampling stations in the Nelson House study area.

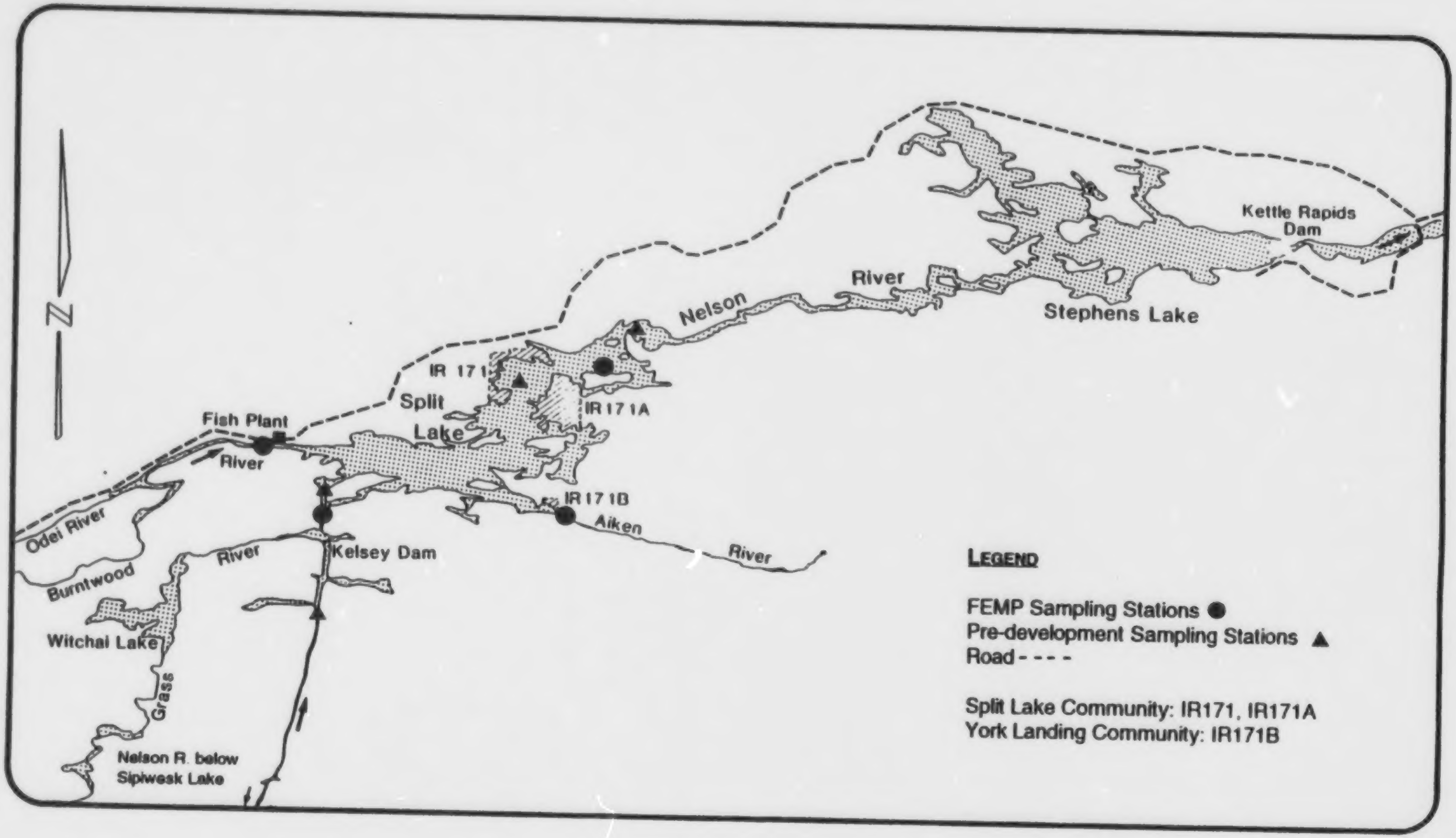
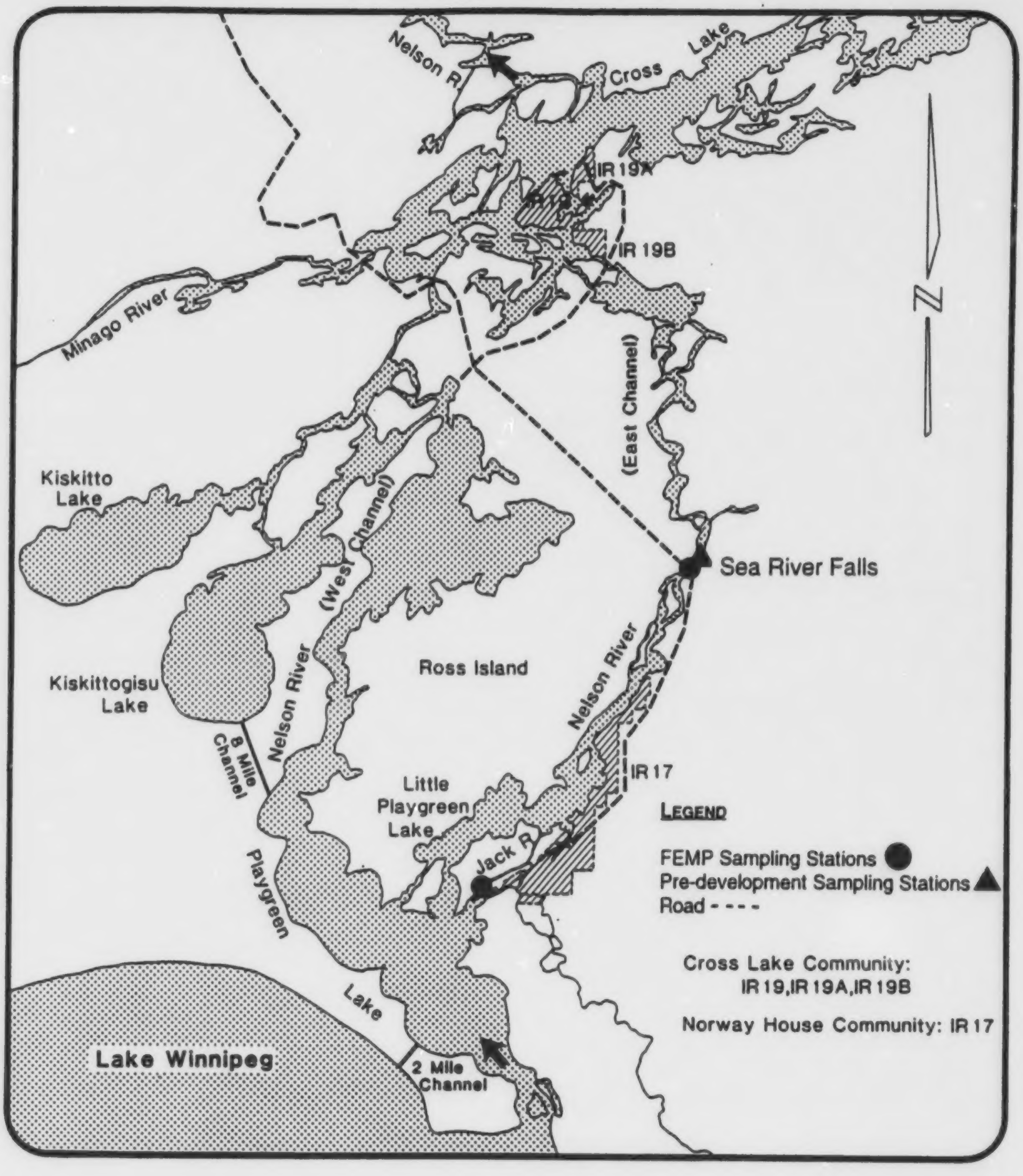

N
Kettle Rapids Dam
Stephens Lake
River
Nelson
IR 171
IR171A
Split
Lake
Fish Plant
River
IR171B
Aiken
River
Odei River
Burntwood
River
Kelsey Dam
Witchai Lake
Grass
Nelson R. below Sipiwesk Lake
LEGEND
FEMP Sampling Stations ●
Pre-development Sampling Stations ▲
Road - - - -
Split Lake Community: IR171, IR171A
York Landing Community: IR171B

Figure 5. Location of FEMP and pre-development water quality sampling stations in the Split Lake study area.



Nelson R.
Cross Lake
IR 19A
IR 19B
N
Minago River
(East Channel)
Kiskitto Lake
(West Channel)
Sea River Falls
Ross Island
Kiskittogisu Lake
Nelson River
Nelson River
8 Mile Channel
IR 17
Little Playgreen Lake
Jack R.
Playgreen Lake
Lake Winnipeg
2 Mile Channel
LEGEND
FEMP Sampling Stations ●
Pre-development Sampling Stations ▲
Road - - - -
Cross Lake Community: IR 19, IR 19A, IR 19B
Norway House Community: IR 17

Figure 6. Location of the FEMP and pre-development water quality sampling stations in the Norway House study area.

All sites were sampled monthly except during freeze-up in November and break-up in April. Nine of the 11 sites were sampled twice monthly from June to September in 1987 and 1988. Although samples were collected from 5 of the sites in September, 1986, routine sampling did not begin at most sites until January, 1987. None of the 1986 data was included in the analysis. Sampling at most sites continued until September-October, 1989. Exceptions to this sampling plan were the sites on the Burntwood River above Threepoint Lake and Footprint River above Footprint Lake where sampling began in May 1987 and continued to March 1989. Details of the sampling procedures may be found in Strange (1991). The variables measured in this program, and the corresponding NAQUADAT analysis codes are listed in Table 1.

2.1.2 Historical Data Sources and Data Screening

Pre-development data for the sites were obtained from a number of sources (Table 2). All data, including those gathered in the FEMP study, were screened for inclusion in the analysis of differences between the pre- and post-development periods to ensure that any significant differences were not the product of differences in analytical methods or errors in analyses or data reporting, and that sufficient data were available for application of the statistical test. Data which were included in the analysis had the following characteristics:

Table 1. Description of water quality variables and abbreviations listed in data tables, including the NAQUADAT analysis codes for the FEMP study data.

Abbreviation	Variables (units)	NAQUADAT Code
pH (F)	pH (Field)	10301F
pH (L)	pH (Lab)	10301L
Spec Cond (F)	Specific Conductance at 25°C (Field) ($\mu S\ cm^{-1}$)	02041F
Spec Cond (L)	Specific Conductance at 25°C (Lab) ($\mu S\ cm^{-1}$)	02041L
Turbidity (F)	Turbidity (Field) (NTU)	02073F
Turbidity (L)	Turbidity (Lab) (NTU)	02081L
Colour	True Colour (Relative units)	02021L
TDS	Total Dissolved Solids (mg L^{-1})	00201L
NFR	Non-filterable Residue (mg L^{-1})	10401L
NFR Fixed	Non-filterable, Fixed Residue (mg L^{-1})	10501L
Tot Alk	Total Alkalinity, as $CaCO_3$ (mg L^{-1})	10111L
Bicarb	Bicarbonate Alkalinity (mg L^{-1})	06201L
Tot Hard	Total Hardness, as $CaCO_3$ (mg L^{-1})	10602L
Non-carb Hard	Non-carbonate Hardness (mg L^{-1})	10650L
Free CO_2	Free Carbon Dioxide (mg L^{-1})	06401L
Part P	Particulate Phosphorus (mg L^{-1})	15901L
Ortho P	Ortho Phosphorus (mg L^{-1})	15256D
TDP	Total Dissolved Phosphorus (mg L^{-1})	15103D
Tot P	Total Phosphorus (mg L^{-1})	15406L
Part N	Particulate Nitrogen (mg L^{-1})	07901L
TDN	Total Dissolved Nitrogen (mg L^{-1})	07651L
Diss NO_2NO_3	Dissolved Nitrite and Nitrate (mg L^{-1})	07110D
Tot NH_3	Total Ammonia, as N (mg L^{-1})	07540P
Tot N	Total Nitrogen (mg L^{-1})	07603L
DOC	Dissolved Organic Carbon (mg L^{-1})	06104L
POC	Particulate Organic Carbon (mg L^{-1})	06901L
TOC	Total Organic Carbon (mg L^{-1})	06002L
DIC	Dissolved Inorganic Carbon (mg L^{-1})	06181L
Chl a	Chlorophyll a (mg L^{-1})	06717G
Diss Ca	Dissolved Calcium (mg L^{-1})	20103L
Diss Mg	Dissolved Magnesium (mg L^{-1})	12102L

Table 1. continued.

Abbreviation	Parameters	NAQUADAT Codes
Diss K	Dissolved Potassium (mg L^{-1})	19103L
Diss Na	Dissolved Sodium (mg L^{-1})	11103L
React Si	Reactive Silica, as SiO_2(mg L^{-1})	14102L
Diss Cl	Dissolved Chloride (mg L^{-1})	17206L
Diss F	Dissolved Flouride (mg L^{-1})	09106L
Diss SO_4	Dissolved Sulphate (mg L^{-1})	16306L
Extr Al	Extractable Aluminum (mg L^{-1})	13302P
Diss Al	Dissolved Aluminum (mg L^{-1})	13102G
Diss As	Dissolved Arsenic (mg L^{-1})	33108D
Tot Ba	Total Barium (mg L^{-1})	56001P
Tot Cd	Total Cadmium (mg L^{-1})	48002P
Diss Cd	Dissolved Cadmium (mg L^{-1})	48102G
Tot Co	Total Cobalt (mg L^{-1})	27002P
Tot Cu	Total Copper (mg L^{-1})	29005P
Diss Cu	Dissolved Copper (mg L^{-1})	29105G
Extr Fe	Extractable Iron (mg L^{-1})	26304P
Diss Fe	Dissolved Iron (mg L^{-1})	26104G
Tot Pb	Total Lead (mg L^{-1})	82002P
Diss Pb	Dissolved Lead (mg L^{-1})	82103G
Extr Mn	Extractable Manganese (mg L^{-1})	25304P
Diss Mn	Dissolved Manganese (mg L^{-1})	25104G
Tot Hg	Total Mercury (mg L^{-1})	80011P
Tot Ni	Total Nickel (mg L^{-1})	28002P
Diss Ni	Dissolved Nickel (mg L^{-1})	28102G
Diss Se	Dissolved Selenium (mg L^{-1})	34108D
Tot V	Total Vanadium (mg L^{-1})	23002P
Diss V	Dissolved Vanadium (mg L^{-1})	23102G
Tot Zn	Total Zinc (mg L^{-1})	30005P
Diss Zn	Dissolved Zinc (mg L^{-1})	30105G

Table 2. Sources of pre-development data used in the analysis of water quality changes.

Site	Source Station Name (No.)	Period of Record Used	Variables Used
Churchill R. near Leaf Rapids	NAQUADAT MA06EB0002	July 1972 to August 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, Diss Na, Diss K, React Si
Rat R. above Threepoint L.	NAQUADAT MA05TF0001	March 1972 to October 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, NO_2NO_3, Ortho P, Tot P, Diss Na, Diss K, Diss Ca, Diss Cl, Diss F, Diss SO_4, React Si, Diss Fe, Diss Mn, Diss Cu, Diss Pb, Diss Zn
Footprint R. above Footprint L.	NAQUADAT MA05TF0003	March 1972 to December 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, NO_2NO_3, Ortho P, Tot P, Diss Na, Diss K, Diss Ca, Diss Cl, Diss F, Diss SO_4, React Si, Diss Fe, Diss Mn, Diss Cu, Diss Pb, Diss Zn
Burntwood R. above Threepoint L.	NAQUADAT MA05TF0002	March 1972 to December 1973	pH (L), Spec Cond (L), Turbidity (L), Tot Alk, Tot Hard, TOC, NO_2NO_3, Ortho P, Tot P, Diss Na, Diss K, Diss Ca, Diss Cl, Diss F, Diss SO_4, React Si, Diss Fe, Diss Mn, Diss Cu, Diss Pb, Diss Zn
Burntwood R. at Thompson	NAQUADAT MA05TG0001	February 1967 to December 1977	pH (L), Spec Cond (L), Turbidity (L), NFR, Tot Alk, Tot Hard, Tot P, Diss NO_2NO_3, TOC, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss F, Diss SO_4, Extr Al, Extr Fe, Diss Fe, Extr Mn
	Morelli (1974) Burntwood R. at Thompson		Tot P, Diss Ca, Diss Cl, Diss SO_4
Nelson R. near outlet to Split L.	NAQUADAT* Split L. at Split L.	July 1972 to October 1973	pH (L), Spec Cond (L), Turbidity (L), Colour, NFR, NFR Fixed, Tot Alk, Tot Hard, Diss Ca, Diss K, Diss Na, React Si, Diss Cl, Diss F, Ortho P, Tot P, Diss NO_2NO_3, TOC, Diss Cu, Diss Fe, Diss Pb, Diss Mn, Diss Zn
	Cleugh (1974) and Hecky & Harper (1974) Split L. at Outlet	June 1972 to September 1973	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO_4, Diss Fe, Diss Mn
Nelson R. near inlet to Split L.	Cleugh (1974) Nelson R. below Sipiwesk L.	February to December 1973	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO_4, Diss Fe, Diss Mn
	Cleugh (1974) Nelson R. above Kelsey Dam	May to October 1972	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO_4, Diss Fe, Diss Mn
	Cleugh (1974) Nelson R. below Kelsey Dam	May 1972 to September 1973	pH (L), Spec Cond (F), TDS, NFR, TDP, Susp P, TDN, Susp N, DOC, Susp C, Chla a, Diss Ca, Diss Mg, Diss K, Diss Na, React Si, Diss Cl, Diss SO_4, Diss Fe, Diss Mn
Nelson R. below Sea River Falls	NAQUADAT MA05UB0006	June 1971 to May 1972	pH (L), Spec Cond (L), Turbidity (L), NFR, NFR Fixed, Tot Alk, Tot Hard, Diss NO_2NO_3, TOC, Diss K, Diss Na, React Si, Diss Cu, Diss Pb, Diss Zn

* Contains data from Morelli (1974)

- were the product of comparable analytical methods in the pre-and post-development periods;

- included measurements in at least 2 calender years (either at the same site or at closely located replicate sites) before and 2 calender years after development with at least 3 measurements in each year and with samples in both the ice-free and ice-cover seasons (ensures enough data to screen outlying values);

- no more than 20% of the measurements in a period were below the analytical limit of detection (avoids use of mean values which are below detection); and,

- all outlying values (i.e., unusually high or low relative to other measurements for the period) which could not be confirmed as valid were removed.

Based on these criteria, the minimum dataset which could be included in the analysis consisted of 3 samples (either 2 ice-free and 1 ice-cover or vice-versa). In practice, the historical datasets included in the analysis generally consisted of 5 to 9 samples with 4 to 6 of the samples during the ice-free season.

In cases where data used were below the analytical limit of detection, they were assigned the value of 50% of the limit of detection following the convention employed by Playle and

Williamson (1988).

The comparability of analytical methods was assessed in a number of ways. For the data from the NAQUADAT database, this comparison was relatively straightforward as the analytical methods were identified as parameter codes in the data records. Measurements for which the parameter codes have not changed were considered to be directly comparable. The consequence of any change in parameter code during the period of record on data comparability was investigated further with officials of Environment Canada. If there was any doubt regarding comparability for any parameter, that parameter was excluded from consideration. The analyses of changes in water quality in the Churchill River above Leaf Rapids and in the Footprint, Rat, and Burntwood rivers above Threepoint Lake relied almost exclusively on NAQUADAT data. This provided some assurance that any significant changes detected in the analysis were not due to a change in analytical method. Where possible, this approach was verified by examining temporal changes in the natural rivers. This analysis (see Section 3.1.5) revealed that dramatic changes in SO_4 and NO_2-NO_3 concentrations occurred in the natural rivers between the pre-development and FEMP study periods. In the absence of any identifiable cause, this suggested there have been some undocumented changes in sampling or analytical methods or in analytical laboratories.

The analyses of changes in water quality in the Split Lake area relied on data from a number of sources including NAQUADAT and the LWCNR reports (Cleugh 1974; Hecky and Harper 1974). The analytical methods used in these studies were reviewed and compared with those employed in the current study and found to use the same basic procedures (i.e., temperature correction of pH and conductivity values; major ions and dissolved metals by atomic absorption). In the absence of parallel measurements it is not possible to declare the methods identical, but our investigation indicated the data should be suitable for detection of changes of the magnitude to which the system has been subjected (i.e., ± 25-50% of baseline values).

Although total Hg concentrations were measured in water samples from all 11 sites, these data were not included in the analysis due to inadequacies in the sampling and analytical methods. The detection limits of the method ranged from 0.01 to 0.02 $\mu g\ L^{-1}$ over the course of the study, a level which has been shown to be inadequate for the ambient concentrations. Actual levels on the CRD and Nelson River range between 0.001 and 0.003 $\mu g\ L^{-1}$ (Ramsey 1990). The sampling procedures employed in the FEMP study also were not adequate for the prevention of sample contamination, as indicated by the occurrence of measurable concentrations at a detection limit which was a factor of 10 higher than actual levels.

2.2 Statistical Analysis

2.2.1 Water Quality - Water Discharge Relationships

At riverine sites, there often is a correlation between the measured value of physical and chemical water quality variables and river discharge as has been found at a number of sites in northern Manitoba (Duncan and Williamson 1988). Examination of these relationships can provide general information on the nature of impacts related to modification of river flow regimes.

A negative slope, the result of decreasing concentrations with increasing discharge, indicates the primary effect of higher flows is dilution of the substance in question. In this circumstance, the increased flows either do not increase the supply of a substance to the watercourse at all, or that supply does not increase as quickly as flow. Negative slopes for dissolved constituents are characteristic of streams in which baseflow is maintained by sub-surface drainage through the watershed (Hynes 1970). Sub-surface runoff percolates through the soils, picking up solutes along the way. Thus, water quality in receiving streams is a reflection of soil and bedrock composition in the watershed and integrates local variations in chemical composition. Surface flows predominate during storm events or spring run-off. The run-off water is in contact with the soils for a shorter period, resulting in lower solute concentrations. This produces decreasing chemical concentrations with increasing discharge and the corresponding negative correlation.

A positive slope, the product of increasing concentrations with increasing flow, indicates the primary effect of higher flows is to concentrate the substance in question. Particulate constituents (e.g., NFR, turbidity, POC) are often positively correlated with discharge. This can occur if higher flows cause the introduction of the substance to the water column through bank and shoreline erosion or through surficial erosion within the watershed.

Relationships with discharge also can be used to reduce the magnitude of variation among samples in a period of interest, permitting a greater sensitivity for the detection of significant differences between periods. Knowledge of the relationship with discharge can also be used as an impact prediction and management tool to determine the impact of proposed changes in the discharge regimes of control and generation structures as required by the addition of generating capacity to the Churchill-Nelson project.

The individual water quality parameters were regressed on daily discharge for each period for each site. A linear model was selected as there was no reason to consider a higher order polynomial. In a similar analysis which included several of the sites examined in the present study, Duncan and Williamson (1988) found a 2nd or 3rd order polynomial did not significantly improve the fit of the models.

The data record for each site was stratified according to the sequence of hydroelectric development. These periods are indicated in Table 3. The analysis was performed for each period of the data record to examine changes in the relationship which might be related to the hydroelectric project. If data were collected from more than one location during a period for a given site, as indicated in Table 2, the regression analysis was conducted separately for each location. A correlation was considered

Table 3. Stratification of historical water quality data records by phase of construction. Dates of construction start and completion from G. McCullough, DFO, Winnipeg.

Site	Period Name	Period of Record
Burntwood R. above Threepoint L.	Pre-development	March 1972 to December 1973
Rat R. above Threepoint L.	Pre-development	March 1972 to December 1973
Footprint R. above Footprint L.	Pre-development	March 1972 to December 1973
Nelson R. below Sea River Falls	Pre-development	June 1971 to May 1972
Nelson R. at the inlet to Split L.	Pre-development	May 1972 to September 1973
Nelson R. at the outlet to Split L.	Pre-development	July 1972 to October 1973
Burntwood R. at Thompson	Pre-development	February 1967 to May 1974
	Construction	June 1974 to August 1976
	Immediate Post-diversion	September 1976 to December 1977

significant at $p \leq 0.05$. Sources of discharge data which were used in the analysis are listed in Table 4.

Variables which were significantly correlated with discharge during a period at a given site were standardized to the mean discharge for that period at that site as follows:

$$^{std}[Y]_i = [\hat{Y}]_{ref} - ([\hat{Y}]_i - [Y]_i)$$

where, for a given site:

$^{std}[Y]_i$ is the concentration of Y in the i^{th} sample standardized to mean discharge;

$[Y]_i$ is the observed concentration of Y in the i^{th} sample;

$[\hat{Y}]_i$ is the predicted concentration of Y at the observed mean daily discharge (Q) on the day of the i^{th} sample; and,

$[\hat{Y}]_{ref}$ is the predicted concentration of Y at the mean Q for the period calculated as:

$$[\hat{Y}]_{ref} = m \cdot Q_{ref} + b$$

where:

Q_{ref} is the mean discharge for the period at the site; and,

m and b are the slope and intercept, respectively, of the regression relationship between Y and Q.

This discharge-standardization procedure is depicted graphically in Figure 7.

Standardization to discharge eliminates the variance in Y accounted for by the relationship with Q, but has no effect on

Table 4. Sources of discharge data used in the water quality/water discharge analysis.

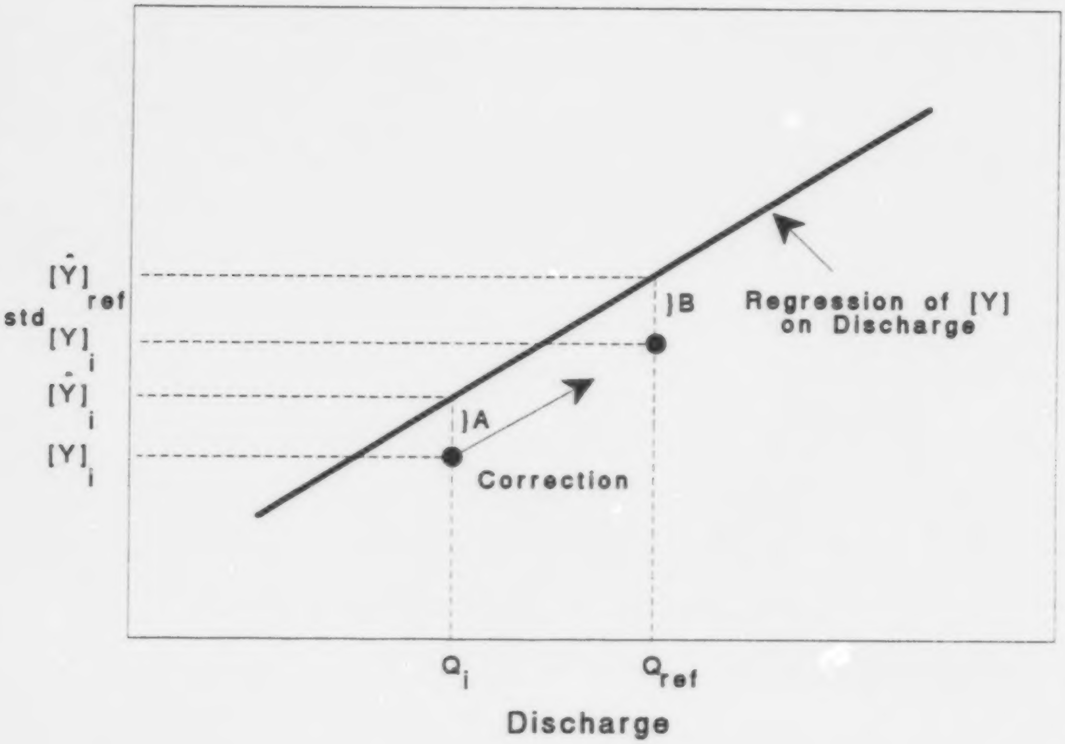
Site	Information Source	Sites Used (W.S.C. codes)	
		Measured Discharge	Calculated Discharge
Jack R. (W. Channel) above Norway House	No Data		
Nelson R. below Sea River Falls	W.S.C.	05UB008	
Burntwood R. near Thompson	W.S.C.	05TG001	
Burntwood R. above Threepoint L.	W.S.C.	05TE002	
Rat R. above Threepoint L.	Man. Hydro	05TF003	
Footprint R. above Footprint L.	W.S.C.	05TF002	
Burntwood R. at outlet of Threepoint L.	Man. Hydro/W.S.C.		05TE002 + 05TF003 + 05TF002
Burntwood R. at inlet to Split L.	W.S.C.		05TG001 + 05TG003[1]
Nelson R. below confluence with Grass R.	Man. Hydro/W.S.C.		05UE005[2] + 05TD001[3]
Aiken R. at inlet to Split L.	No Data		
Nelson R. at outlet of Split L.	Man. Hydro/W.S.C.	05UF006	

W.S.C. Water Survey of Canada (1990)

[1] Odei R. near Thompson

[2] Nelson R. at Kelsey Generating Station

[3] Grass R. above Standing Stone Falls


$[\hat{Y}]_{ref}$
$^{std}[Y]_i$
$[\hat{Y}]_i$
$[Y]_i$
}B
}A
Correction
Regression of [Y] on Discharge
Q_i
Q_{ref}
Discharge

Where: $A = [\hat{Y}]_i - [Y]_i$

$B = [\hat{Y}]_{ref} - {}^{std}[Y]_i$

Let: $A = B$

Therefore: ${}^{std}[Y]_i = [\hat{Y}]_{ref} - ([\hat{Y}]_i - [Y]_i)$

Figure 7. Graphical representation of the procedure used to standardize chemistry data to discharge. See text for further definition of terms. Figure modified from Jones et al. (1985).

other sources of variance in Y, thereby reducing the total variance in Y. Reduction of the total variance in Y can improve the sensitivity of an analysis of change in Y over time because the smaller the variance in Y, the smaller the change necessary for it to be statistically significant (Snedecor and Cochran 1980). This approach is commonly used in the analysis of temporal trends in mercury levels in fish tissues in order to remove the influence of covariates such as fish size or age (Ramsey 1991).

Discharge-standardized values, when a significant correlation with discharge indicated they could be calculated, were used in the analysis of changes in water quality between the pre- and post-development periods. The use of standardized values has been indicated in all tables. None of the figures contained in this report presents standardized values.

2.2.2 Analysis of Changes in Water Quality

For the Burntwood, Rat, and Footprint rivers above Threepoint Lake, the Nelson River at the inlet to Split Lake, and the outlet of Split Lake, the water quality data records amount to a short period immediately before the start of construction (1972-1973) and the data collected in the FEMP study (Table 3). The significance of changes in water quality at these sites was examined using Student's t-test (Snedecor and Cochran 1980). Replicates were the annual mean values for each variable. In most cases, successive years of data from the same source (e.g., NAQUADAT; Table 2) comprised the replicates for a period. However, replicates for the pre-development period both at the Nelson River inlet to Split Lake and in the vicinity of the Split Lake outlet were comprised of data from several sources (Table 2) in order to provide a sufficient number of replicates for analysis. In those instances where data from more than one location were used, the annual mean values for each location were treated as individual replicates.

The data record for the Burntwood River at Thompson is more extensive and was stratified into four periods (Table 3). The significance of changes in water quality at this site was examined using one-way analysis of variance (ANOVA; Snedecor and Cochran 1980) followed by the Tukey/HSD multiple comparison test on all possible pairwise comparisons (Wilkinson 1988).

The use of discharge-standardized values in the analysis of temporal changes is comparable to the use of analysis of covariance (Snedecor and Cochran 1980), with discharge as the covariate. Covariance analysis could have been used in this study, however, standardization to discharge followed by ANOVA or a t-test was found to be more expeditious.

Data were screened for violations of the assumption of homogeneity of variance using Bartlett's test (Snedecor and Cochran 1980). When such violations occurred, the variance usually could be stabilized with a log-transformation although a square-root transformation was occasionally necessary. Notably, inhomogeneity of variance never caused a false significant difference. Stabilization of the variance by transformation generally caused the significance of a difference to increase rather than decrease.

The use of annual means as replicates differs from the approach used in previous analyses of water quality changes in the NFA area by Playle and Williamson (1986) and Guilbault et al. (1979). In these studies, the individual measurements were used as replicates. Annual means were used in the present study to ensure independence of the experimental units (Lettenmaier 1976).

All data manipulations and statistical analyses were performed on an IBM compatible 386 computer using SYSTAT Version 4.0 (Wilkinson 1988).

2.2.3 Seasonal Differences

One of the strong points of the FEMP water quality study was the collection of samples under ice-cover as well as during the ice-free season. This permits analysis of current differences in water quality between the ice-cover and ice-free seasons and evaluation of the significance of changes in seasonal patterns related to the hydroelectric developments at sites where sufficient pre-development data are available.

The comprehensive statistical analysis of significant seasonal differences in a water quality database of the size considered in this study is a substantial task. In order to reduce the analysis to a manageable size, the data were screened for the probable occurrence of seasonal differences, as indicated by winter measurements falling outside the range of summer measurements. All occurrences of seasonal differences revealed by this screening process, both in the pre-development and FEMP datasets, are identified in the report along with graphs illustrating the timing and magnitude of these differences. Seasonal data are not presented for variables in which seasonal differences were not evident based on the screening criteria.

No statistical analysis of any apparent changes in seasonal water quality patterns between the pre- and post-development periods was undertaken due to the general deficiency of the pre-development datasets. None of the sites where initial screening indicated changes in the seasonal water quality patterns may have occurred had a sufficient number of pre-development measurements (i.e., at least 3 in a season and at least 2 years of data both pre- and post-development) for application of a statistical analysis.

2.3 Bacteriological Water Quality

The bacteriological water quality study sampled recreational waters at all 5 NFA communities. At Cross Lake, Norway House, Split Lake, and Nelson House the studies were comprised of two components, a seasonal sampling program and an intensive sampling program. Only the seasonal sampling program was conducted at York Landing. Although a number of bacterial parameters (fecal and total coliform and fecal streptococcus) were measured in these studies, only the results of the fecal coliform counts are presented in this report. The fecal coliform bacteria count is the only parameter for which a water quality guideline has been established.

The seasonal sampling program involved the collection of single samples from 7 to 12 widely separated stations near each community on five occasions between late May and late September, 1987. Stations included popular and remote beach areas, raw water sources for water treatment plant intakes, and sewage treatment plant outfalls. Station numbers, descriptions, and locations are listed in Appendix 1.

The seasonal sampling program was conducted to provide a general indication of bacteriological water quality in each of the communities. A major deficiency of this program was its non-compliance with the sampling frequency specified in the Health and Welfare Canada Guidelines for Recreational Water Quality of at least 5 samples within a 30 day period (Health and Welfare Canada 1989). The sites were sampled on 5 occasions, but over a period of almost 120 days. Thus, the results of the seasonal study program can not be used to determine if recreational water quality is in strict compliance with the guideline. Rather, they provide a general measure of bacteriological water quality and can be used to identify sites requiring further examination.

The intensive sampling program involved the collection of 10 to 11 samples from each of 1 or 2 sites in the communities of Norway House, Cross Lake, Split Lake, and Nelson House on 2 dates in August, 1987. Sampling sites were located near sewage treatment plant outfalls and water treatment plant intakes. In Cross Lake, sampling was concentrated within 150 m of the water treatment plant intake on both sampling dates. In Norway House, 7 of the 11 samples on August 11 and 7 of the 10 samples on August 24 were collected within 180 m of the sewage treatment plant outfall with the remaining samples on both dates from the vicinity of the water treatment plant. Station numbers, descriptions and locations, and sampling and analytical methods are listed in Appendix 1.

2.3.1 Data Interpretation

The suitability of water at the study sites for recreational use was assessed by comparison with the Health and Welfare Canada (1989) Guideline for Recreational Water Quality which specifies the geometric mean fecal coliform bacteria count should be lower than 200 bacteria per 100 mL. The geometric mean is the antilog of the arithmetic mean of the $\log_{10}$-transformed bacterial counts (Elliot 1977). This guideline has been established to minimize the risk of swimmers developing gastroenteritis. Epidemiological studies have shown that the risk of contracting gastroenteritis is about 0.6% at a fecal coliform count of 200 bacteria per 100 mL (McGovern 1983).

As indicated in Section 2.2, the geometric mean fecal coliform counts from the seasonal study only provide a general measure of recreational water quality. This sampling frequency might be sufficient for the identification of serious violations of the water quality guideline but is inadequate for assessment of strict compliance. This inadequacy of the seasonal sampling program was alleviated in large measure by the intensive sampling program, which collected up to 20 samples from a single site within a period of 14-21 days. The geometric mean fecal coliform counts from the intensive sampling program can therefore be used to assess compliance with the Health and Welfare Canada guideline. These intensive studies typically were conducted in the vicinity of sewage treatment plant outfalls; sites with the highest potential for bacterial contamination and therefore sites of the greatest concern.

3.0 Results and Discussion

3.1 Natural Rivers

3.1.1 Current Water Quality

Water quality in the three natural, unregulated, rivers (Burntwood R. above Threepoint Lake; Footprint R. above Footprint Lake; Aiken R.) was characterized by a number of variables which exceeded one or more of the Canadian guidelines for drinking water, recreational use, and protection of aquatic life. Mean turbidity values exceeded the drinking water guideline at all three sites (Tables 5, 6, 7, and 8). All measurements, both lab and field, from the Burntwood River exceeded the guideline as did all field measurements from the Footprint River.

The guideline for turbidity has been set to meet both health and aesthetic concerns. Turbidity can affect disinfection efficiency, biological nutrient availability, trihalomethane formation, and can carry toxic heavy metals and biocides (CCREM 1987). It should be kept in mind that the prevalence of turbidity values in excess of the Canadian guideline is not an indicator of pollution, only that the water should be treated for turbidity

Table 5. Canadian water quality guidelines (CCREM 1987) for drinking water, recreational use, and protection of aquatic life.

Variables	Drinking	Recreational	Aquatic Life
pH	6.5-8.5	5.0-9.0	6.5-9.0
Turbidity	5 NTU	<50 NTU	
Colour	15 TCU		
TDS	500 mg L^{-1}		
NFR			increase of 10 mg L^{-1}[a]
Extractable Al			0.005 mg L^{-1} ($pH \leq 6.5$) 0.1 mg L^{-1} ($pH > 6.5$)
Dissolved As	0.05 mg L^{-1}		0.05 mg L^{-1}
Total Ba	1.0 mg L^{-1}		
Total Cd	0.005 mg L^{-1}		0.0002-0.0008 mg L^{-1}[b]
Total Cu	1.0 mg L^{-1}[c]		0.002 mg L^{-1}
Extractable Fe	0.3 mg L^{-1}[c]		0.3 mg L^{-1}
Total Pb	0.05 mg L^{-1}		0.001-0.002 mg L^{-1}[d]
Extractable Mn	0.05 mg L^{-1}		
Total Ni			0.025-0.065 mg L^{-1}[e]
Dissolved Se	0.01 mg L^{-1}		0.001 mg L^{-1}
Total Zn	5.0 mg L^{-1}		0.03 mg L^{-1}
Dissolved NO_2NO_3	10 mg L^{-1}		
Total NH_3			2.2 mg L^{-1}[f] 1.37 mg L^{-1}[g]
Dissolved Cl	250 mg L^{-1}		
Dissolved F	1.5 mg L^{-1}		
Dissolved SO_4	500 mg L^{-1}		

[a] Background NFR ≤100 mg L^{-1}

[b] 0.0002 mg L^{-1} at hardness 0-60 mg L^{-1}
0.0008 mg L^{-1} at hardness 60-120 mg L^{-1}

[c] Aesthetic

[d] 0.001 mg L^{-1} at hardness 0-60 mg L^{-1}
0.002 mg L^{-1} at hardness 60-120 mg L^{-1}

[e] 0.025 mg L^{-1} at hardness 0-60 mg L^{-1}
0.065 mg L^{-1} at hardness 60-120 mg L^{-1}

[f] pH 6.5, 10°C

[g] pH 8.0, 10°C

Table 6. Summary statistics for physical and chemical water quality variables measured in the Burntwood River above Threepoint Lake; May, 1987, to March, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.2	7.9	7.2
	Lab		7.40	0.20	7.97	7.00
Conductivity	Field	$\mu S\ cm^{-1}$	101	22	142	24
	Lab		112	17	167	82
Turbidity	Field	NTU	23 +	7	47 +	12 +
	Lab		19 +	4	27 +	13 +
Colour	True	Relative	41 +	10	55 +	25 +
TDS		mg L^{-1}	57	10	71	25
NFR	Total	mg L^{-1}	14	7	33	6
	Fixed	mg L^{-1}	11	5	27	4
Alkalinity	Total	mg L^{-1}	50.1	6.3	61.4	36.2
	Bicarbonate	mg L^{-1}	63.4	13.1	115.9	44.1
Hardness	Total	mg L^{-1}	51.2	5.6	61.7	39.6
	Non-carbonate	mg L^{-1}	1.5	1.3	3.5	0
Free CO_2		mg L^{-1}	3.32	1.92	6.70	0
Phosphorus	Particulate	mg L^{-1}	0.020	0.011	0.062	<0.006
	Ortho PO_4	mg L^{-1}	0.003	0.004	0.022	<0.003
	Total Dissolved	mg L^{-1}	0.016	0.009	0.050	0.006
	Total	mg L^{-1}	0.036	0.016	0.112	0.021
Nitrogen	Particulate	mg L^{-1}	0.102	0.099	0.529	0.010
	Total Dissolved	mg L^{-1}	0.454	0.104	0.771	0.260
	NO_2-NO_3	mg L^{-1}	0.049	0.059	0.205	<0.010
	Total NH_3	mg L^{-1}	0.035	0.021	0.096	0.012
	Total	mg L^{-1}	0.56	0.14	1.07	0.37
Carbon	DOC	mg L^{-1}	11.96	0.86	14.00	10.40
	POC	mg L^{-1}	0.724	0.470	2.640	0.310
	TOC	mg L^{-1}	12.68	1.13	15.24	10.87
	DIC	mg L^{-1}	11.7	1.8	14.7	8.5
Chlorophyll-a		mg L^{-1}	0.004	0.005	0.022	<0.001
Ca	Dissolved	mg L^{-1}	13.0	1.4	15.8	10.1
Mg	Dissolved	mg L^{-1}	4.6	0.5	5.4	3.5
K	Dissolved	mg L^{-1}	0.96	0.15	1.29	0.75
Na	Dissolved	mg L^{-1}	2.5	0.4	3.3	1.9

Table 6. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	3.25	0.92	5.23	1.69
Cl	Dissolved	mg L^{-1}	1.0	0.2	1.5	0.7
F	Dissolved	mg L^{-1}	0.07	0.01	0.10	0.06
SO_4	Dissolved	mg L^{-1}	2.4	0.7	4.1	1.0
Al	Extractable	mg L^{-1}	0.443 +++	0.148	0.700 +++	0.191 +++
	Dissolved	mg L^{-1}	0.100	0.082	0.443	<0.100
As	Dissolved	mg L^{-1}	0.0003	0.0001	0.0004	0.0001
Ba	Total	mg L^{-1}	<0.08		<0.08	<0.08
Cd	Total	mg L^{-1}	0.0001	0.0002	0.0009 +++	<0.0001
	Dissolved	mg L^{-1}	<0.0001		0.0001	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0003	0.0013	<0.0005
Cu	Total	mg L^{-1}	0.0061 +++	0.0081	0.0296 +++	0.0012
	Dissolved	mg L^{-1}	0.0017	0.0005	0.0026	0.0011
Fe	Extractable	mg L^{-1}	0.642 +, +++	0.165	1.050 +, +++	0.183
	Dissolved	mg L^{-1}	0.142	0.077	0.350	0.035
Pb	Total	mg L^{-1}	0.0009	0.0007	0.0028 +++	<0.0007
	Dissolved	mg L^{-1}	<0.0007		<0.0007	<0.0007
Mn	Extractable	mg L^{-1}	0.041	0.016	0.090 +	0.005
	Dissolved	mg L^{-1}	0.014	0.017	0.056	<0.002
Ni	Total	mg L^{-1}	0.0015	0.0005	0.0026	0.0007
	Dissolved	mg L^{-1}	0.0008	0.0005	0.0018	<0.0005
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0003	<0.0001
V	Total	mg L^{-1}	0.0015	0.0013	0.0070	<0.0005
	Dissolved	mg L^{-1}	0.0005	0.0003	0.0010	<0.0005
Zn	Total	mg L^{-1}	0.0041	0.0033	0.0127	0.0009
	Dissolved	mg L^{-1}	0.0016	0.0010	0.0030	0.0005
Discharge	Daily	m^3 sec^{-1}	17.9	11.8	44.0	3.0

* Exceeds limit for drinking water.

** Exceeds limit for recreational use.

*** Exceeds limit for protection of aquatic life.

Table 7. Summary statistics for physical and chemical water quality variables measured in the Footprint River above Footprint Lake; May, 1987, to March, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.7	0.5	8.5	7.0
	Lab		7.48	0.33	7.87	6.15
Conductivity	Field	$\mu S\ cm^{-1}$	178	38	249	112
	Lab		191	42	268	121
Turbidity	Field	NTU	12^{+}	6	36^{+}	6^{+}
	Lab		9^{+}	3	18^{+}	4
Colour	True	Relative	20^{+}	8	40^{+}	10
TDS		$mg\ L^{-1}$	99	26	147	41
NFR	Total	$mg\ L^{-1}$	8	4	20	4
	Fixed	$mg\ L^{-1}$	6	3	16	2
Alkalinity	Total	$mg\ L^{-1}$	96.0	21.7	138.0	56.8
	Bicarbonate	$mg\ L^{-1}$	117.0	26.5	168.2	69.2
Hardness	Total	$mg\ L^{-1}$	96.2	22.0	139.4	61.2
	Non-carbonate	$mg\ L^{-1}$	1.2	1.7	6.7	0
Free CO_2		$mg\ L^{-1}$	5.39	5.92	22.29	0
Phosphorus	Particulate	$mg\ L^{-1}$	0.018	0.007	0.041	0.007
	Ortho PO_4	$mg\ L^{-1}$	<0.003	0.001	0.003	<0.003
	Total Dissolved	$mg\ L^{-1}$	0.012	0.005	0.031	0.005
	Total	$mg\ L^{-1}$	0.030	0.007	0.051	0.018
Nitrogen	Particulate	$mg\ L^{-1}$	0.093	0.049	0.248	0.040
	Total Dissolved	$mg\ L^{-1}$	0.458	0.069	0.610	0.280
	NO_2-NO_3	$mg\ L^{-1}$	0.022	0.027	0.110	<0.010
	Total NH_3	$mg\ L^{-1}$	0.043	0.031	0.124	0.012
	Total	$mg\ L^{-1}$	0.55	0.10	0.86	0.41
Carbon	DOC	$mg\ L^{-1}$	11.27	0.75	13.00	10.00
	POC	$mg\ L^{-1}$	0.640	0.260	1.270	0.340
	TOC	$mg\ L^{-1}$	11.91	0.78	14.00	10.57
	DIC	$mg\ L^{-1}$	22.8	5.4	33.1	13.6
Chlorophyll-a		$mg\ L^{-1}$	0.004	0.005	0.024	<0.001
Ca	Dissolved	$mg\ L^{-1}$	27.5	6.7	40.5	17.6
Mg	Dissolved	$mg\ L^{-1}$	6.7	1.3	9.3	4.2
K	Dissolved	$mg\ L^{-1}$	0.90	0.23	1.37	0.59
Na	Dissolved	$mg\ L^{-1}$	2.4	0.5	3.5	1.5

Table 7. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	2.82	1.45	5.07	0.25
Cl	Dissolved	mg L^{-1}	1.0	0.3	1.8	0.7
F	Dissolved	mg L^{-1}	0.07	0.01	0.09	0.06
SO_4	Dissolved	mg L^{-1}	1.9	0.7	3.3	<0.2
Al	Extractable	mg L^{-1}	0.190^{+++}	0.097	0.410^{+++}	<0.100
	Dissolved	mg L^{-1}	<0.100	0.029	0.148	<0.100
As	Dissolved	mg L^{-1}	0.0003	0.0001	0.0004	0.0001
Ba	Total	mg L^{-1}	<0.08		<0.08	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L^{-1}	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0002	0.0011	<0.0005
Cu	Total	mg L^{-1}	0.0232^{+++}	0.1060	0.5530^{+++}	0.0005
	Dissolved	mg L^{-1}	0.0011	0.0007	0.0027	0.0005
Fe	Extractable	mg L^{-1}	$0.420^{+,+++}$	0.246	$1.150^{+,+++}$	0.091
	Dissolved	mg L^{-1}	0.123	0.125	0.438	0.010
Pb	Total	mg L^{-1}	0.0011	0.0030	0.0161^{+++}	<0.0007
	Dissolved	mg L^{-1}	<0.0007	0.0001	0.0007	<0.0007
Mn	Extractable	mg L^{-1}	0.057^{+}	0.078	0.296^{+}	<0.002
	Dissolved	mg L^{-1}	0.047	0.064	0.296	<0.002
Ni	Total	mg L^{-1}	0.0009	0.0003	0.0016	<0.0005
	Dissolved	mg L^{-1}	0.0007	0.0003	0.0012	<0.0005
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0002	<0.0001
V	Total	mg L^{-1}	0.0008	0.0004	0.0014	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0002	0.0006	<0.0005
Zn	Total	mg L^{-1}	0.0040	0.0099	0.0515^{+++}	0.0004
	Dissolved	mg L^{-1}	0.0009	0.0010	0.0032	<0.0003
Discharge	Daily	m^3 sec^{-1}	3.0	2.3	9.0	0

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recreational use.

$^{+++}$ Exceeds limit for protection of aquatic life.

Table 8. Summary statistics for physical and chemical water quality in the Aiken River at the inlet to Split Lake, January, 1987, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.3	8.1	6.9
	Lab		7.57	0.28	7.96	6.74
Conductivity	Field	$\mu S\ cm^{-1}$	185	85	360	66
	Lab		196	92	370	75
Turbidity	Field	NTU	7^{+}	3	16^{+}	2
	Lab		6^{+}	4	18^{+}	1
Colour	True	Relative	46^{+}	25	120^{+}	8
TDS		$mg\ L^{-1}$	104	51	212	27
NFR	Total	$mg\ L^{-1}$	5	4	18	<1
	Fixed	$mg\ L^{-1}$	3	2	8	<1
Alkalinity	Total	$mg\ L^{-1}$	84.9	36.4	195.0	32.4
	Bicarbonate	$mg\ L^{-1}$	105.1	43.8	237.7	39.5
Hardness	Total	$mg\ L^{-1}$	91.3	36.5	187.2	39.4
	Non-carbonate	$mg\ L^{-1}$	6.5	4.7	17.3	0
Free CO_2		$mg\ L^{-1}$	5.82	8.85	47.58	0
Phosphorus	Particulate	$mg\ L^{-1}$	0.012	0.009	0.036	<0.006
	Ortho PO_4	$mg\ L^{-1}$	<0.003	0.002	0.010	<0.003
	Total Dissolved	$mg\ L^{-1}$	0.011	0.006	0.030	0.004
	Total	$mg\ L^{-1}$	0.023	0.010	0.060	0.008
Nitrogen	Particulate	$mg\ L^{-1}$	0.067	0.064	0.404	0.020
	Total Dissolved	$mg\ L^{-1}$	0.398	0.087	0.756	0.230
	NO_2-NO_3	$mg\ L^{-1}$	0.011	0.014	0.058	<0.010
	Total NH_3	$mg\ L^{-1}$	0.037	0.066	0.385	0.007
	Total	$mg\ L^{-1}$	0.47	0.14	1.16	0.29
Carbon	DOC	$mg\ L^{-1}$	12.16	4.09	23.10	3.90
	POC	$mg\ L^{-1}$	0.527	0.776	4.900	0.100
	TOC	$mg\ L^{-1}$	12.69	4.50	28.00	4.40
	DIC	$mg\ L^{-1}$	20.3	8.7	46.8	7.8
Chlorophyll-a		$mg\ L^{-1}$	0.003	0.002	0.008	<0.001
Ca	Dissolved	$mg\ L^{-1}$	25.3	9.7	54.7	11.5
Mg	Dissolved	$mg\ L^{-1}$	6.9	3.2	13.2	2.6
K	Dissolved	$mg\ L^{-1}$	0.93	0.68	2.55	0.16
Na	Dissolved	$mg\ L^{-1}$	5.1	5.1	17.9	0.7

Table 8. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	2.27	2.45	13.41	0.45
Cl	Dissolved	mg L^{-1}	5.7	5.8	20.7	0.8
F	Dissolved	mg L^{-1}	0.07	0.03	0.12	0.03
SO_4	Dissolved	mg L^{-1}	7.7	7.9	27.0	1.1
Al	Extractable	mg L^{-1}	0.107^{+++}	0.059	0.226^{+++}	<0.100
	Dissolved	mg L^{-1}	<0.100	0.022	0.153	<0.100
As	Dissolved	mg L^{-1}	0.0003	0.0001	0.0006	0.0001
Ba	Total	mg L^{-1}	<0.08		<0.08	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L^{-1}	<0.0001		0.0001	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0003	0.0022	<0.0005
Cu	Total	mg L^{-1}	0.0029^{+++}	0.0041	0.0170^{+++}	<0.0005
	Dissolved	mg L^{-1}	0.0011	0.0008	0.0030	<0.0005
Fe	Extractable	mg L^{-1}	$0.438^{+,+++}$	1.008	$5.980^{+,+++}$	0.101
	Dissolved	mg L^{-1}	0.268	0.679	5.210	0.017
Pb	Total	mg L^{-1}	<0.0007	0.0003	0.0014	<0.0007
	Dissolved	mg L^{-1}	0.0009	0.0011	0.0036	<0.0007
Mn	Extractable	mg L^{-1}	0.093^{+}	0.339	1.920^{+}	0.005
	Dissolved	mg L^{-1}	0.073	0.292	1.640	<0.002
Ni	Total	mg L^{-1}	0.0005	0.0003	0.0013	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0004	0.0015	<0.0005
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0002	<0.0001
V	Total	mg L^{-1}	<0.0005	0.0003	0.0013	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0001	0.0006	<0.0005
Zn	Total	mg L^{-1}	0.0030	0.0038	0.0166	<0.0003
	Dissolved	mg L^{-1}	0.0012	0.0014	0.0041	<0.0003

[+] Exceeds limit for drinking water.

[++] Exceeds limit for recreational use.

[+++] Exceeds limit for protection of aquatic life.

reduction before distribution in a municipal water system.

Colour also generally exceeded the Canadian guideline for human consumption (Tables 6, 7, and 8). Mean colour in the Footprint River was about 30% above the guideline, while Aiken and Burntwood river colour values were about 3 times the guideline level.

The drinking water guideline for colour was set primarily for aesthetic reasons, with the value of 15 TCU representing the lowest concentration in a glass of water which is detectable to the eye (CCREM 1987). Control of colour also is indirectly related to protection of human health, as the chemicals which are responsible for production of the colour (e.g., humic and fulvic acids) can also carry toxic materials such as heavy metals.

Mean concentrations of extractable Al and Fe and total Cu exceeded the guideline for protection of aquatic life in all 3 natural rivers (Tables 6, 7, and 8). The maximum total Pb concentrations exceeded the aquatic life guideline in the Burntwood and Footprint rivers as did peak total Zn in the Footprint River. The consistent violation of the aquatic life guidelines for Al, Fe, and Cu, and the occasional violation of the guidelines for Pb and Zn therefore appear to be a natural characteristic of these local rivers. The mean extractable Fe concentration also exceeded the drinking water guideline in all three rivers, as did mean extractable Mn in the Aiken and Footprint rivers and maximum extractable Mn in the Burntwood River.

3.1.2 Variation Among the Rivers

There were no significant differences among the three rivers in the annual mean concentrations of many variables, including: most nutrient species (particulate P, ortho PO_4, TDP, total P, particulate N, TDN, total N, DOC, POC, and TOC), most major ions (Mg, K, Na, Cl, SO_4, and F), pH, dissolved As and Se, and several metal species (total Cu and Zn, total Pb, dissolved Fe, and extractable and dissolved Mn; Table 9). The Burntwood River differed from the Footprint and Aiken rivers in having lower conductivity, TDS, hardness, DIC, and Ca, and higher turbidity and extractable Al. Burntwood River NFR also was marginally higher than in the Aiken River.

The substances which are responsible for the colour of a stream also can carry heavy metals (CCREM 1987). Both the Burntwood and Aiken rivers were highly coloured relative to the Footprint River, yet none of the differences in metals among the rivers was accounted for by this difference in colour. Levels of extractable Fe and total Ni and V were highest in the Burntwood River and lowest in the Aiken River (Table 9).

3.1.3 Water Quality - Water Discharge Relationships

Water-quality discharge relationships were examined for just two of the three natural rivers, and then only for the FEMP study period. There are no discharge data for the Aiken River and discharge data for the natural Burntwood and Footprint river sites are available only for the post-development period.

Table 9. Evaluation of the significance of differences in selected water quality parameters among the Aiken River at the inlet to Split Lake, the Burntwood River above Threepoint Lake and the Footprint River above Footprint Lake. Values listed are the mean and standard deviation (s.d.) for the respective site in 1987 and 1988, and the p value from the Tukey/HSD pairwise comparison.

pH (Lab)

	Mean	s.d.	Aiken	BWR
Aiken	7.59	0.06		
BWR	7.41	0.06	0.110	
FPR	7.51	0.05	0.465	0.343

Conductivity (Lab)

	Mean	s.d.	Aiken	BWR
Aiken	170	18		
BWR	112.0^1	4	0.066	
FPR	179.0^1	19	0.855	0.047

Turbidity (Lab)

	Mean	s.d.	Aiken	BWR
Aiken	6	<1		
BWR	19	3	0.014	
FPR	9	<1	0.353	0.031

Colour (True)

	Mean	s.d.	Aiken	BWR
Aiken	45.0	6		
BWR	42.0^1	2	0.896	
FPR	19.0^1	6	0.030	0.039

TDS

	Mean	s.d.	Aiken	BWR
Aiken	90	5		
BWR	57.0^1	3	0.013	
FPR	93.0^1	6	0.888	0.011

NFR (Lab)

	Mean	s.d.	Aiken	BWR
Aiken	4	<1		
BWR	14.0^1	4	0.061	
FPR	9.0^1	2	0.270	0.280

Alkalinity (Total)

	Mean	s.d.	Aiken	BWR
Aiken	75.3	0.4		
BWR	50.0^1	1.6	0.013	
FPR	90.2^1	6.1	0.054	0.004

Hardness (Total)

	Mean	s.d.	Aiken	BWR
Aiken	81.5	4.2		
BWR	51.1^1	0.8	0.028	
FPR	90.1^1	9.1	0.419	0.014

Part P

	Mean	s.d.	Aiken	BWR
Aiken	0.011	0.002		
BWR	0.020	0.003	0.071	
FPR	0.017	0.003	0.170	0.554

Table 9. continued.

Ortho P	Mean	s.d.	Aiken	BWR
Aiken	0.003	0.001		
BWR	0.003	0.001	0.856	
FPR	0.002	<0.001	0.856	0.581

TDP	Mean	s.d.	Aiken	BWR
Aiken	0.010	0.001		
BWR	0.016	0.004	0.200	
FPR	0.013	0.001	0.648	0.469

Total P	Mean	s.d.	Aiken	BWR
Aiken	0.021	0.004		
BWR	0.036	0.007	0.092	
FPR	0.030	0.002	0.282	0.454

Part N	Mean	s.d.	Aiken	BWR
Aiken	0.060	0.002		
BWR	0.105	0.027	0.135	
FPR	0.097	0.009	0.208	0.869

TDN	Mean	s.d.	Aiken	BWR
Aiken	0.390	0.017		
BWR	0.459^1	0.033	0.209	
FPR	0.453^1	0.037	0.248	0.979

Total N	Mean	s.d.	Aiken	BWR
Aiken	0.45	0.02		
BWR	0.56	0.10	0.322	
FPR	0.55^1	0.03	0.366	0.989

DOC	Mean	s.d.	Aiken	BWR
Aiken	11.55	0.92		
BWR	12.06^1	0.36	0.695	
FPR	11.15	0.21	0.788	0.388

POC	Mean	s.d.	Aiken	BWR
Aiken	0.404	0.023		
BWR	0.735	0.141	0.069	
FPR	0.650	0.060	0.139	0.650

TOC	Mean	s.d.	Aiken	BWR
Aiken	11.98	0.94		
BWR	12.78^1	0.16	0.433	
FPR	11.77	0.17	0.930	0.311

Table 9. continued.

DIC

	Mean	s.d.	Aiken	BWR
Aiken	18.0	0.9		
BWR	11.7^1	0.1	0.031	
FPR	21.3^1	2.0	0.148	0.010

Chlorophyll a

	Mean	s.d.	Aiken	BWR
Aiken	0.003	<0.001		
BWR	0.005	<0.001	0.002	
FPR	0.005	<0.001	0.002	1.000

Diss Ca

	Mean	s.d.	Aiken	BWR
Aiken	22.7	0.6		
BWR	13.0^1	0.2	0.015	
FPR	25.8	2.5	0.238	0.007

Diss Mg

	Mean	s.d.	Aiken	BWR
Aiken	6.1	0.6		
BWR	4.6^1	0.1	0.109	
FPR	6.3^1	0.6	0.874	0.076

Diss K

	Mean	s.d.	Aiken	BWR
Aiken	0.73	0.20		
BWR	0.94^1	0.08	0.382	
FPR	0.85	0.08	0.688	0.786

Diss Na

	Mean	s.d.	Aiken	BWR
Aiken	4.0	1.5		
BWR	2.6^1	0.1	0.363	
FPR	2.3^1	0.1	0.280	0.955

React Si

	Mean	s.d.	Aiken	BWR
Aiken	1.69	0.01		
BWR	3.16^1	0.17	0.033	
FPR	2.49	0.50	0.147	0.214

Diss Cl

	Mean	s.d.	Aiken	BWR
Aiken	4.3	1.8		
BWR	1.0^1	<0.1	0.096	
FPR	0.9	0.1	0.091	0.996

Diss F

	Mean	s.d.	Aiken	BWR
Aiken	0.07	<0.01		
BWR	0.07	<0.01	0.682	
FPR	0.07^1	<0.01	1.000	0.682

Table 9. continued.

Diss SO_4

	Mean	s.d.	Aiken	BWR
Aiken	6.2	3.1		
BWR	2.4	0.2	0.241	
FPR	1.8	0.2	0.176	0.932

Extr Al

	Mean	s.d.	Aiken	BWR
Aiken	0.105	0.002		
BWR	0.466	0.014	0.002	
FPR	0.195	0.045	0.088	0.004

Diss Al

	Mean	s.d.	Aiken	BWR
Aiken	0.057	0.003		
BWR	0.099	0.032	0.210	
FPR	0.060	0.003	0.986	0.241

Diss As

	Mean	s.d.	Aiken	BWR
Aiken	0.0003	0.0001		
BWR	0.0003	0.0001	0.540	
FPR	0.0003[1]	<0.0001	0.540	1.000

Tot Cu

	Mean	s.d.	Aiken	BWR
Aiken	0.0029	0.0032		
BWR	0.0055	0.0064	0.989	
FPR	0.0209	0.0346	0.578	0.649

Extr Fe

	Mean	s.d.	Aiken	BWR
Aiken	0.215	0.001		
BWR	0.655	0.058	0.002	
FPR	0.349	0.016	0.061	0.006

Diss Fe

	Mean	s.d.	Aiken	BWR
Aiken	0.087	0.001		
BWR	0.132[1]	0.030	0.366	
FPR	0.085	0.037	0.997	0.343

Tot Pb

	Mean	s.d.	Aiken	BWR
Aiken	0.001	<0.001		
BWR	0.001[1]	<0.001	1.000	
FPR	0.011	0.015	0.798	0.798

Extr Mn

	Mean	s.d.	Aiken	BWR
Aiken	0.016	<0.001		
BWR	0.043[1]	0.004	0.071	
FPR	0.031	0.012	0.269	0.343

Table 9. continued.

Diss Mn

	Mean	s.d.	Aiken	BWR
Aiken	0.006	0.001		
BWR	0.014[1]	0.006	0.824	
FPR	0.021	0.023	0.586	0.890

Tot Ni

	Mean	s.d.	Aiken	BWR
Aiken	0.0004	0.0003		
BWR	0.0015	0.0003	0.001	
FPR	0.0009	0.0001	0.011	0.004

Diss Se

	Mean	s.d.	Aiken	BWR
Aiken	0.0001	<0.0001		
BWR	0.0001	0.0001	1.000	
FPR	0.0001	<0.0001	1.000	1.000

Tot V

	Mean	s.d.	Aiken	BWR
Aiken	0.0004	0.0001		
BWR	0.0013	0.0011	0.015	
FPR	0.0009	0.0004	0.080	0.105

Tot Zn

	Mean	s.d.	Aiken	BWR
Aiken	0.0032	0.0019		
BWR	0.0040	0.0021	0.796	
FPR	0.0036	0.0045	0.723	0.989

---

[1] Discharge-standardized variables.

General characteristics of the discharge relationships in the natural rivers included relatively large numbers of discharge correlated variables, the dominance of negative slopes, and the frequent occurrence of moderate to strong correlations (i.e., r^2 >0.4). Colour, turbidity, NFR, and particulate nutrient species typically were positively correlated with discharge. Conductivity, major ions, related variables such as alkalinity and hardness, metals and dissolved nutrients generally were negatively correlated. Specific characteristics of the discharge relationships are discussed below.

3.1.3.1 Burntwood River above Threepoint Lake

More variables (25) were correlated with discharge at this site than at any of the other sites in the study (Figure 8). Negative correlates included conductivity (field and lab), TDS, alkalinity (total and bicarbonate), hardness (total), TDN, dissolved NO_2-NO_3, total NH_3, DIC, major ions (Ca, Mg, K, Na, Si, Cl), and metals (dissolved Fe, dissolved and extractable Mn). Positive correlates included turbidity (field), colour, NFR (total and fixed), DOC, and TOC. Coefficients of determination (r^2) ranged from <0.2 to >0.8 with 11 of the regressions on discharge accounting for at least 40% of the variance in the dependent variables.

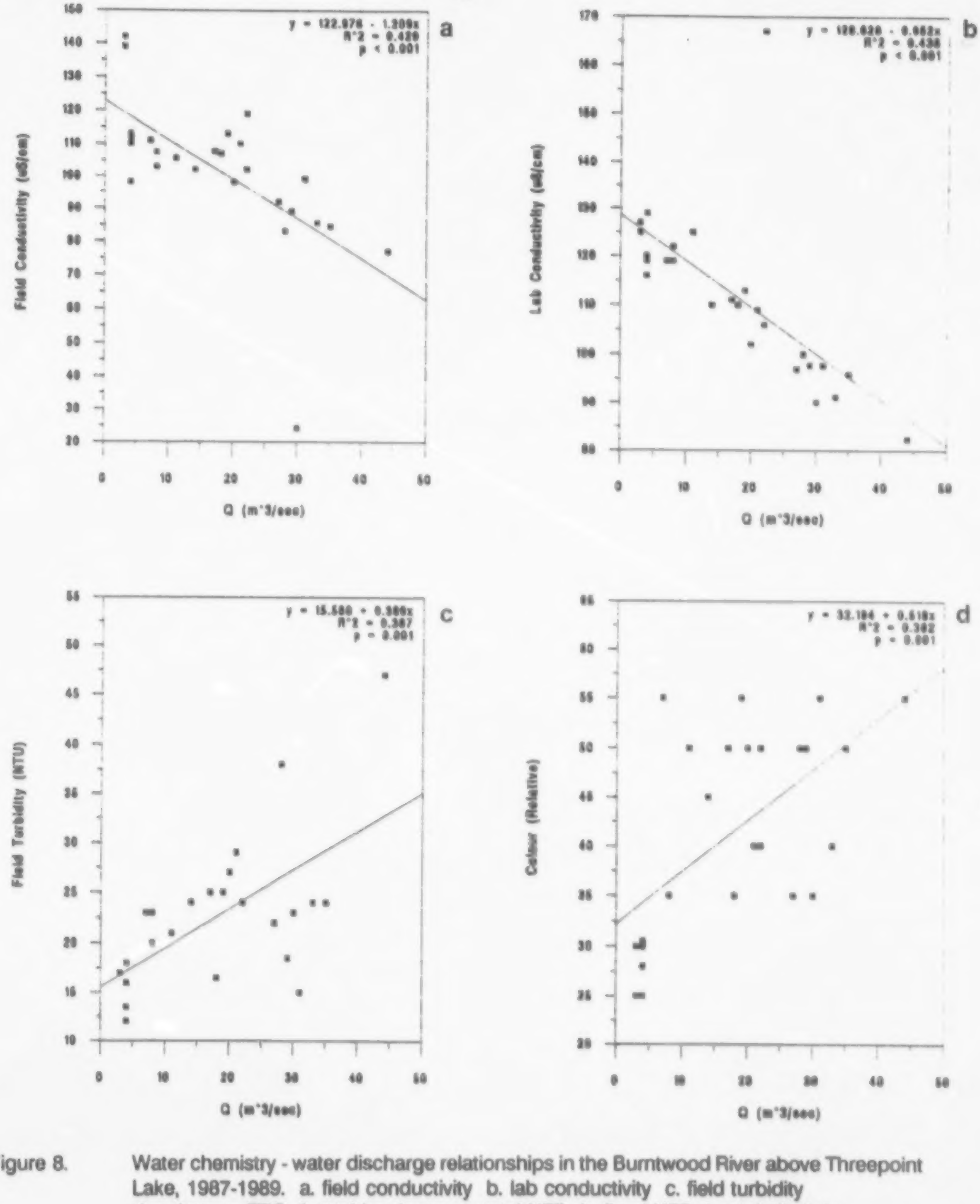
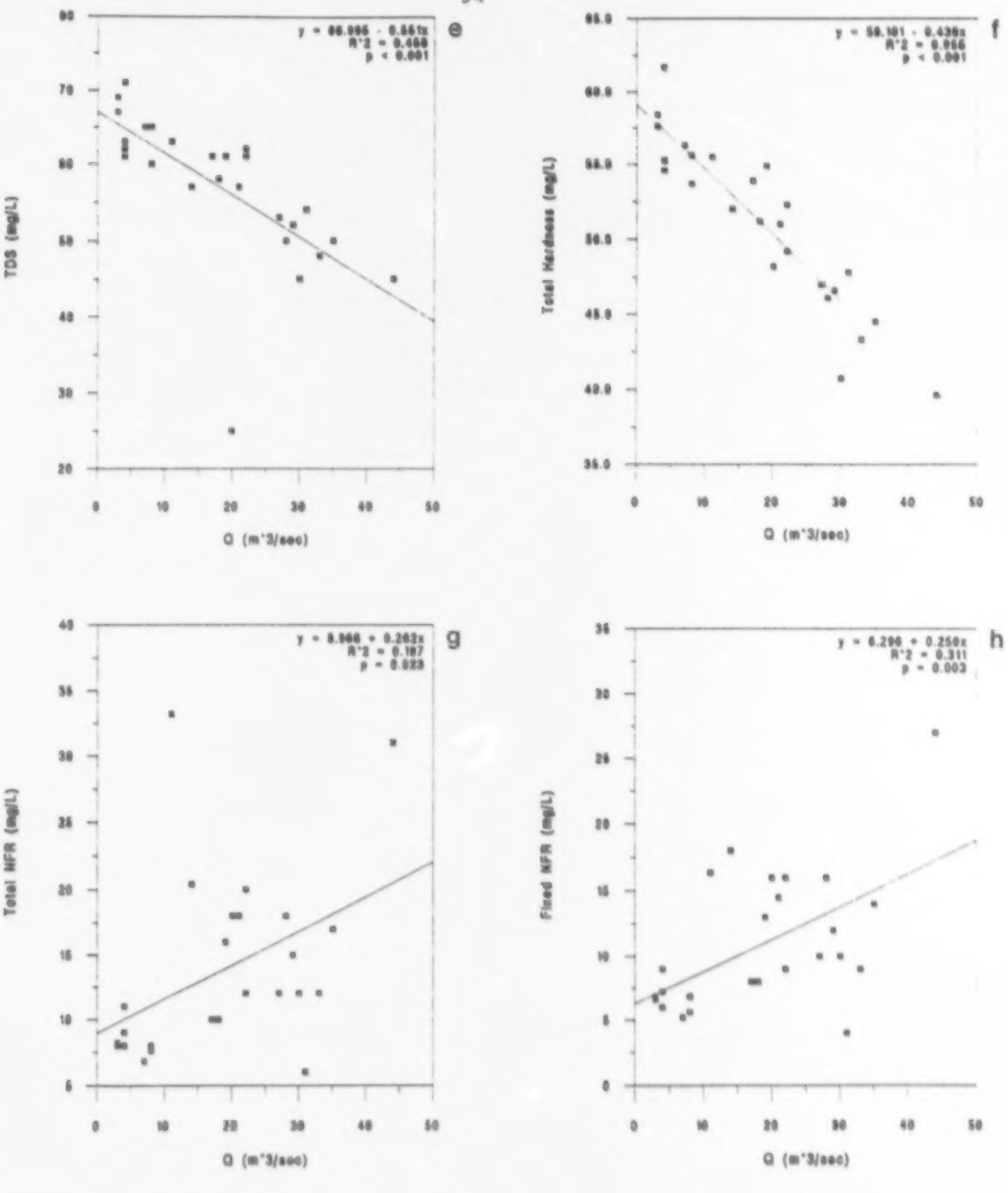


Figure 8. Water chemistry - water discharge relationships in the Burntwood River above Threepoint Lake, 1987-1989. a. field conductivity b. lab conductivity c. field turbidity d. colour e. TDS f. total hardness g. total NFR h. fixed NFR i. total alkalinity j. bicarbonate alkalinity k. TDN l. dissolved NO_2-NO_3 m. total NH_3 n. DOC o. TOC p. DIC q. dissolved Ca r. dissolved Mg s. dissolved K t. dissolved Na u. reactive Si v. dissolved Cl w. dissolved Fe x. extractable Mn y. dissolved Mn.



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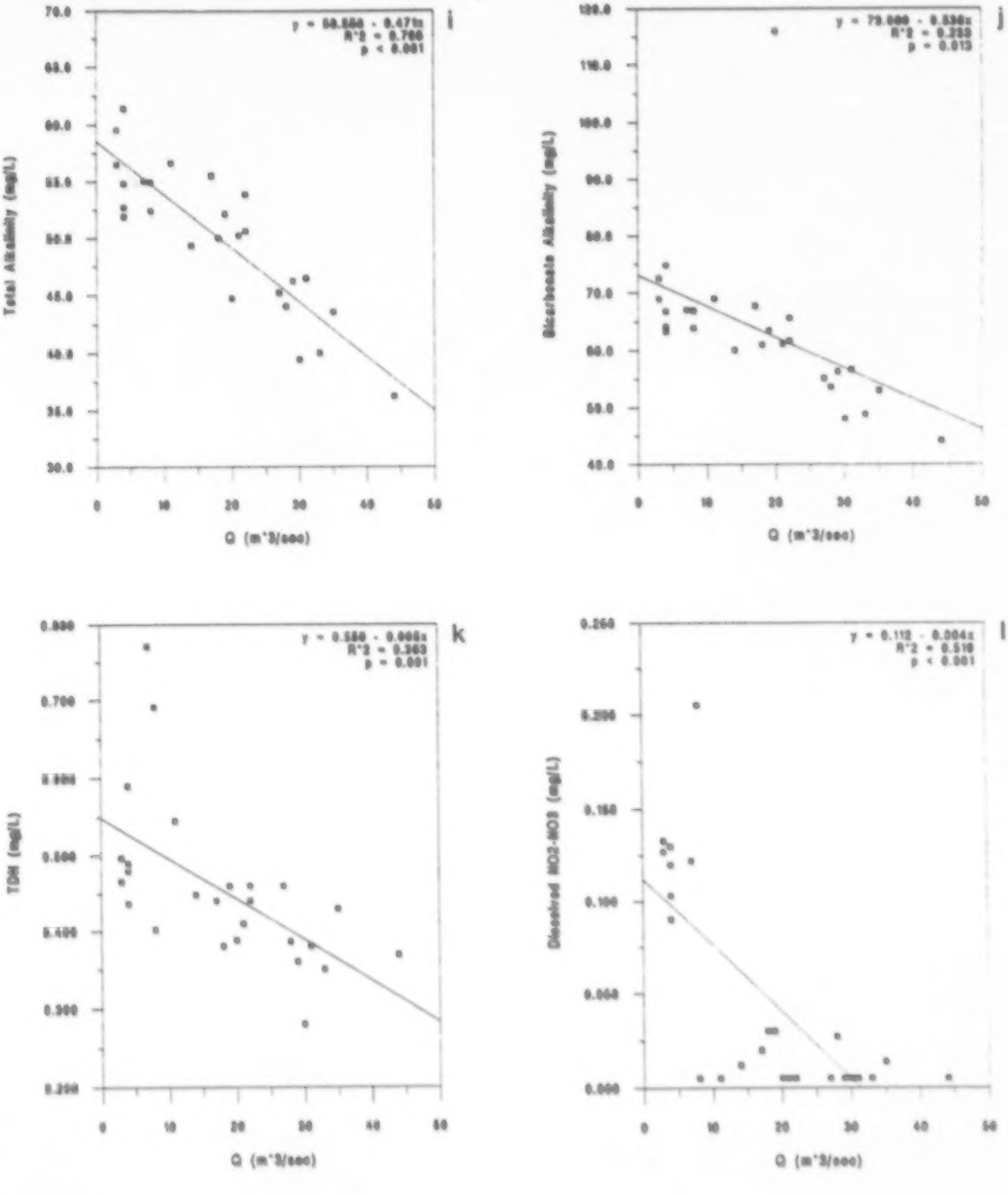

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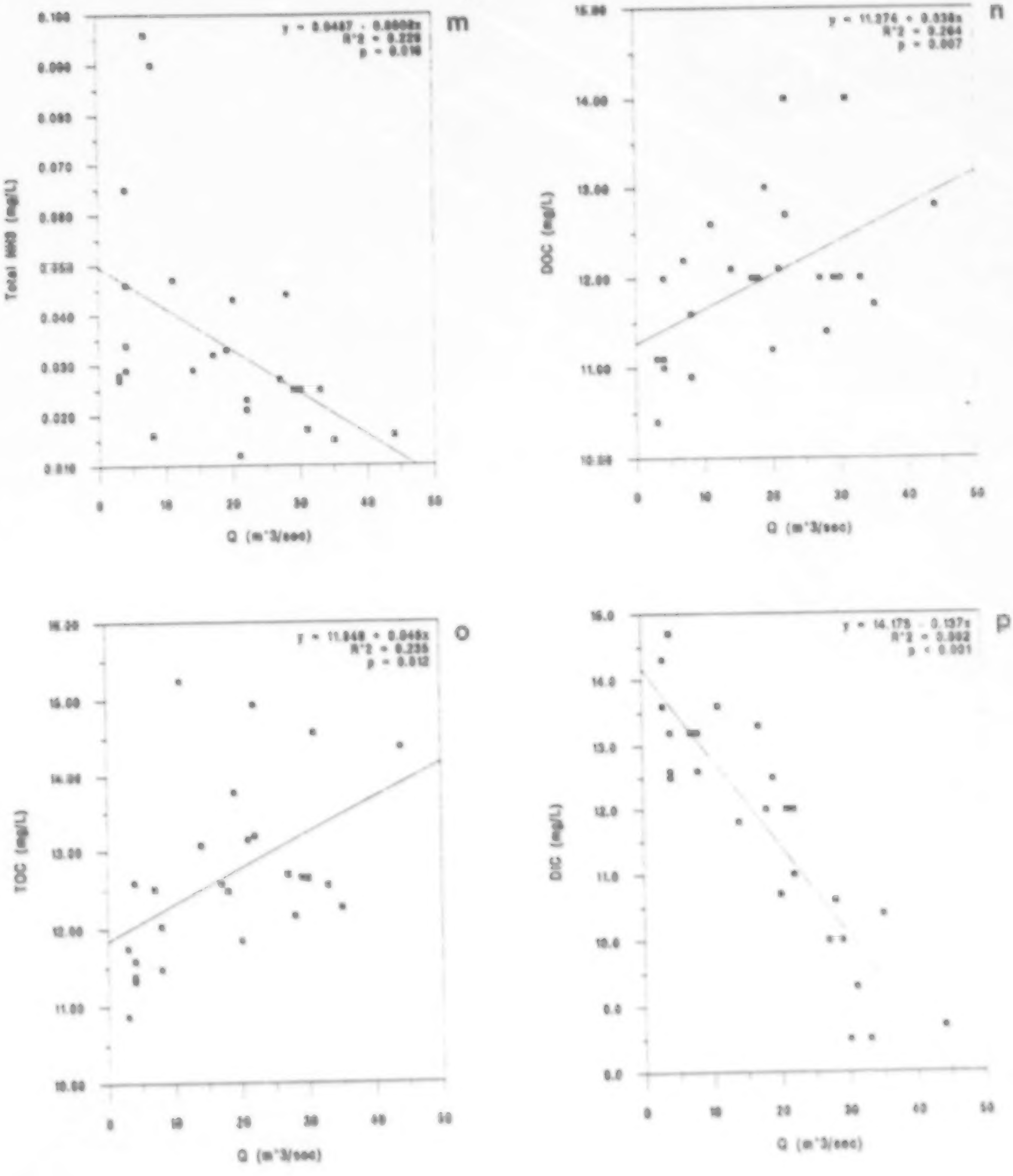

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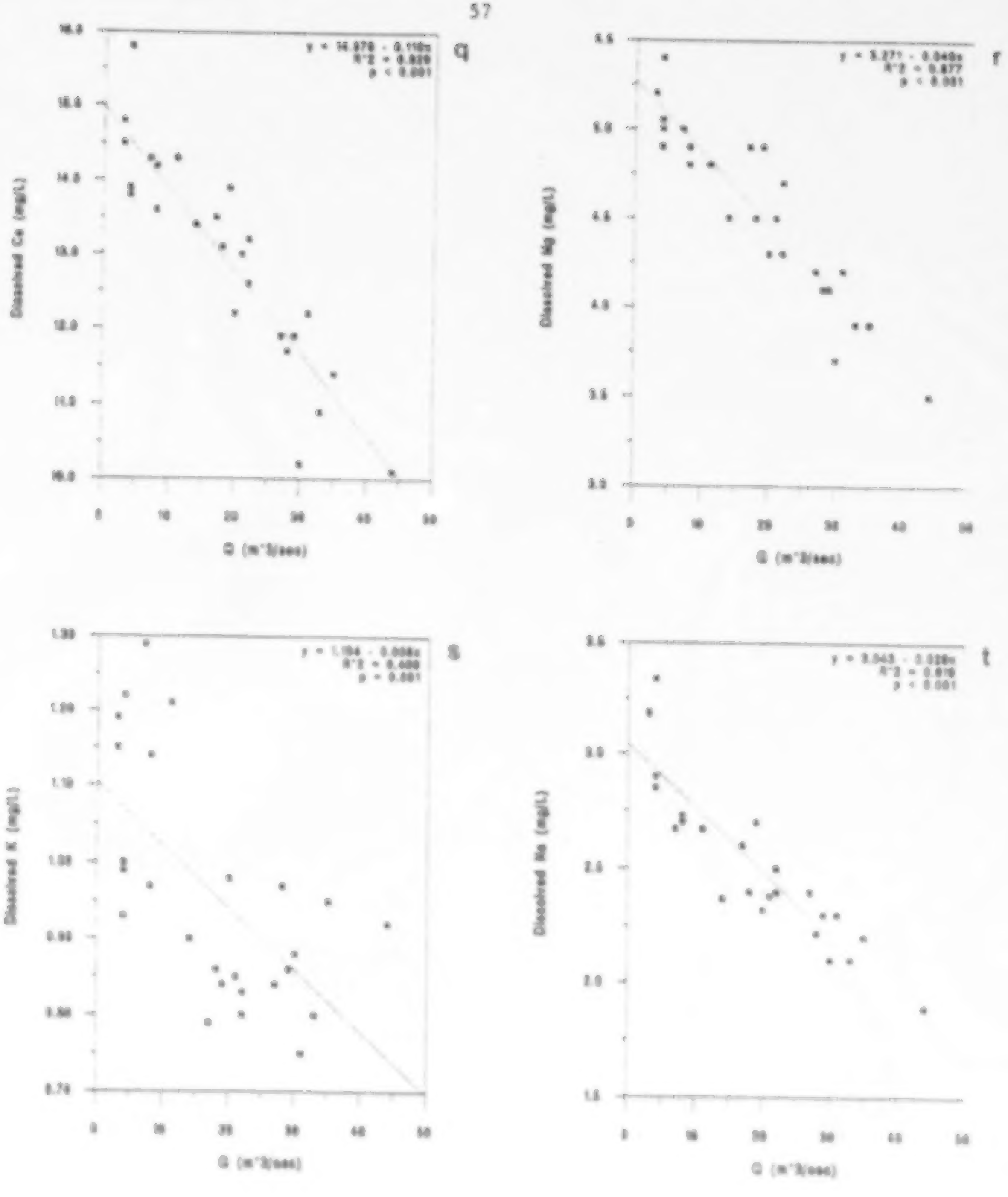

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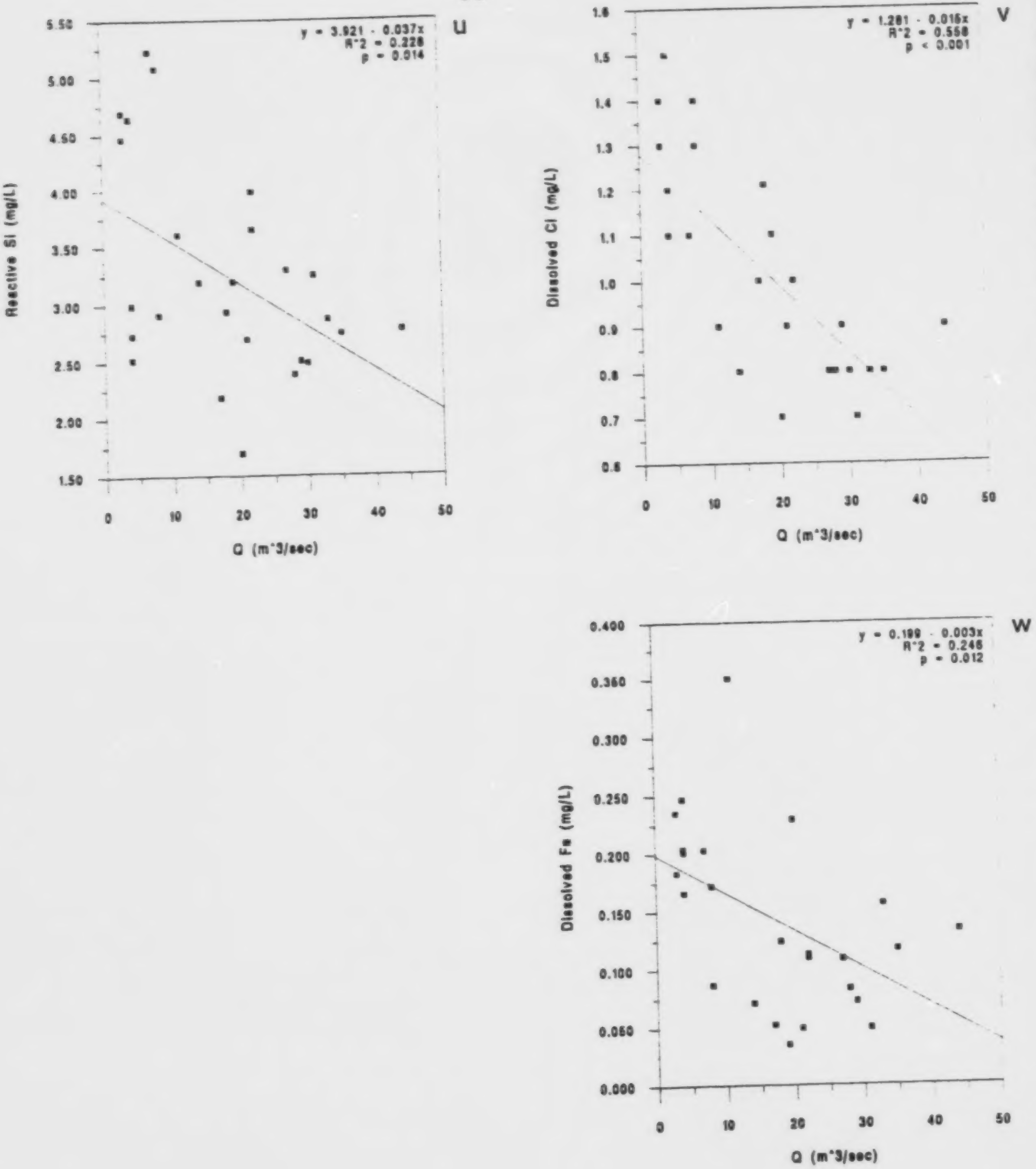

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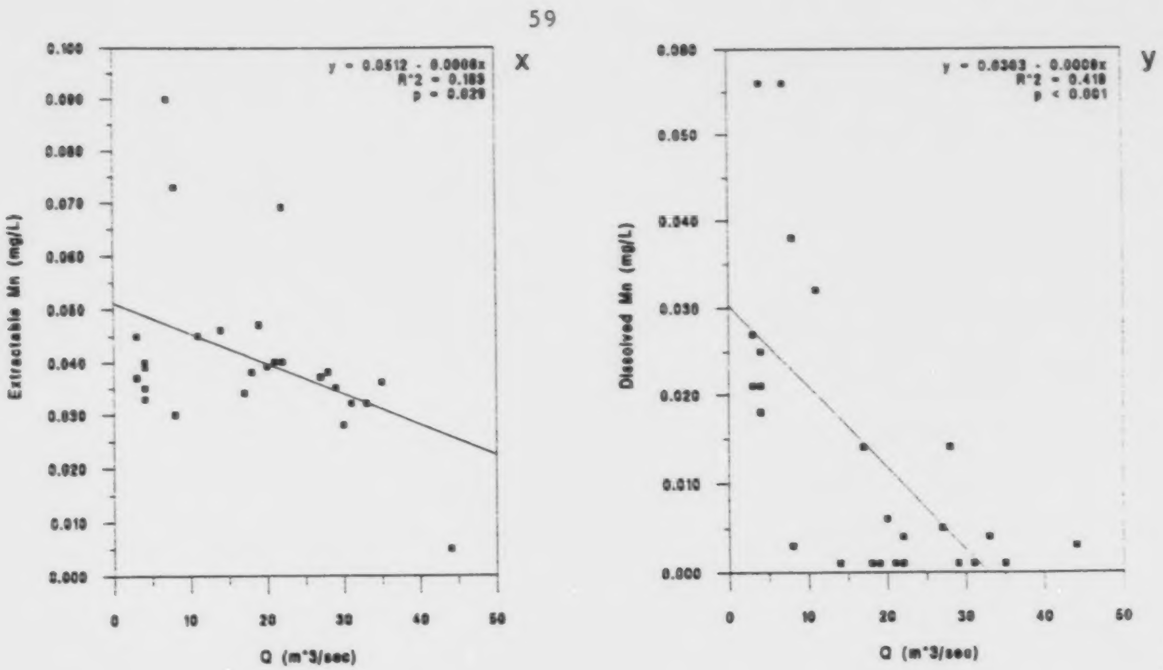


Figure 8. Continued.

3.1.3.2 Footprint River above Footprint Lake

Most of the 17 significant correlations with discharge had a negative slope, as in the Burntwood River above Threepoint Lake (Figure 9). The only positive correlates were colour and NFR (total and fixed). There was a marginally significant positive correlation with turbidity (lab) as well. Field turbidity was not correlated with discharge (r^2=0.089, p=0.085, n=24). Negative correlates included conductivity (field and lab), TDS, alkalinity (total and bicarbonate), hardness (total), TDN, total N, DIC, some major ions (Mg, Na, F), dissolved Ni, and dissolved As. Coefficients of determination ranged from <0.2 to >0.8, with most between 0.15 and 0.40.

3.1.4 Seasonal Differences

3.1.4.1 Burntwood River above Threepoint Lake

There were few differences in water quality between the ice-free and ice-cover seasons in the Burntwood River above Threepoint Lake. Colour and chlorophyll a were lower under ice-cover while concentrations of dissolved Mg, Na, and Zn were higher under ice-cover than during the open water season (Figure 10). The lower chlorophyll a under ice is probably due to the imposition of light limitation on algal growth by accumulated snowcover.

The pre-development dataset for this site included just one winter of 4 measurements (Dec. 1972, and Jan., Feb., Mar, 1973). All of the measurements made under ice-cover during this pre-development study period were within the range for the open water

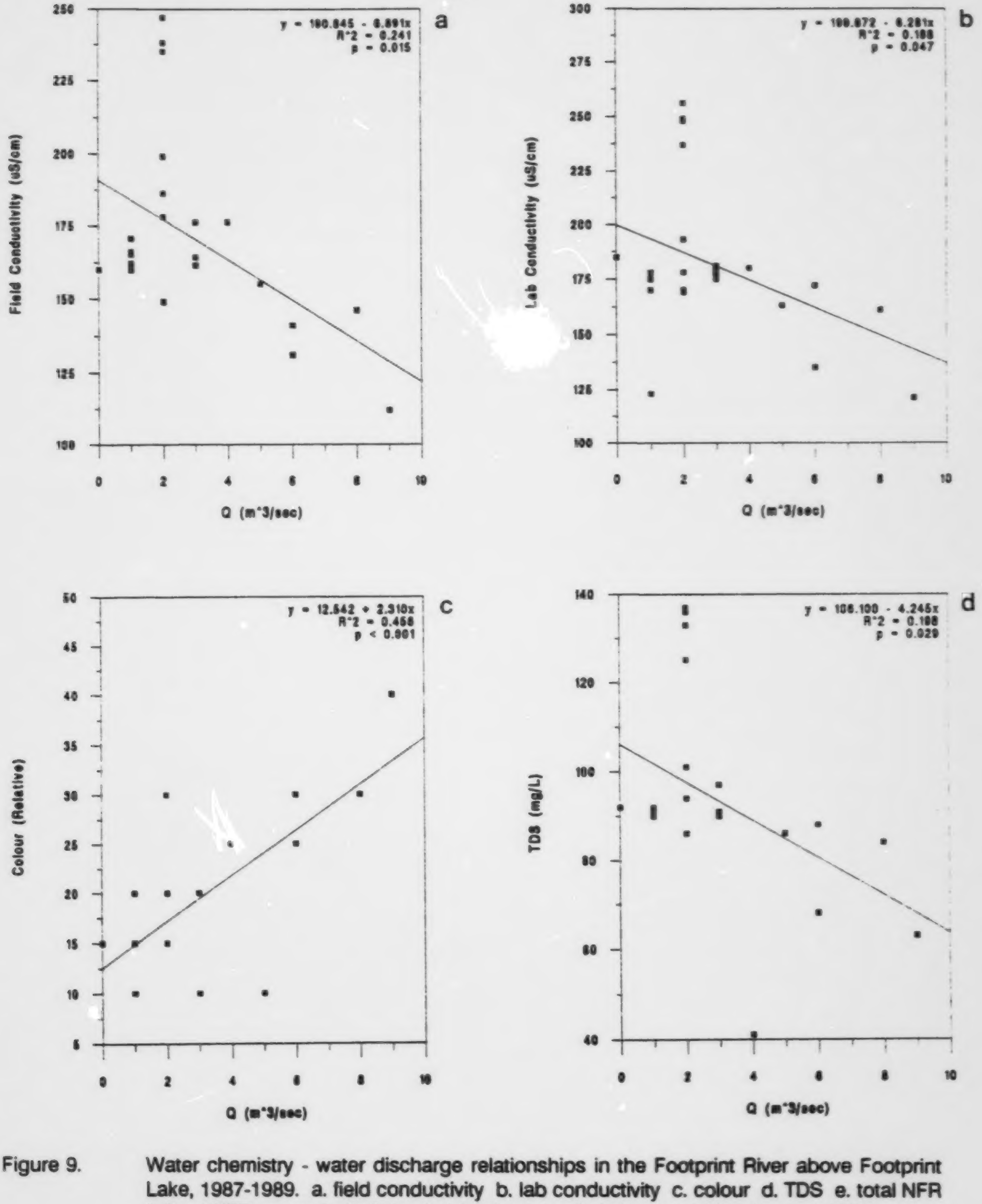


Figure 9. Water chemistry - water discharge relationships in the Footprint River above Footprint Lake, 1987-1989. a. field conductivity b. lab conductivity c. colour d. TDS e. total NFR f. fixed NFR g. total alkalinity h. bicarbonate alkalinity i. total hardness j. TDN k. total N l. DIC m. dissolved Mg n. dissolved Na o. dissolved F p. dissolved As q. dissolved Ni r. lab turbidity.

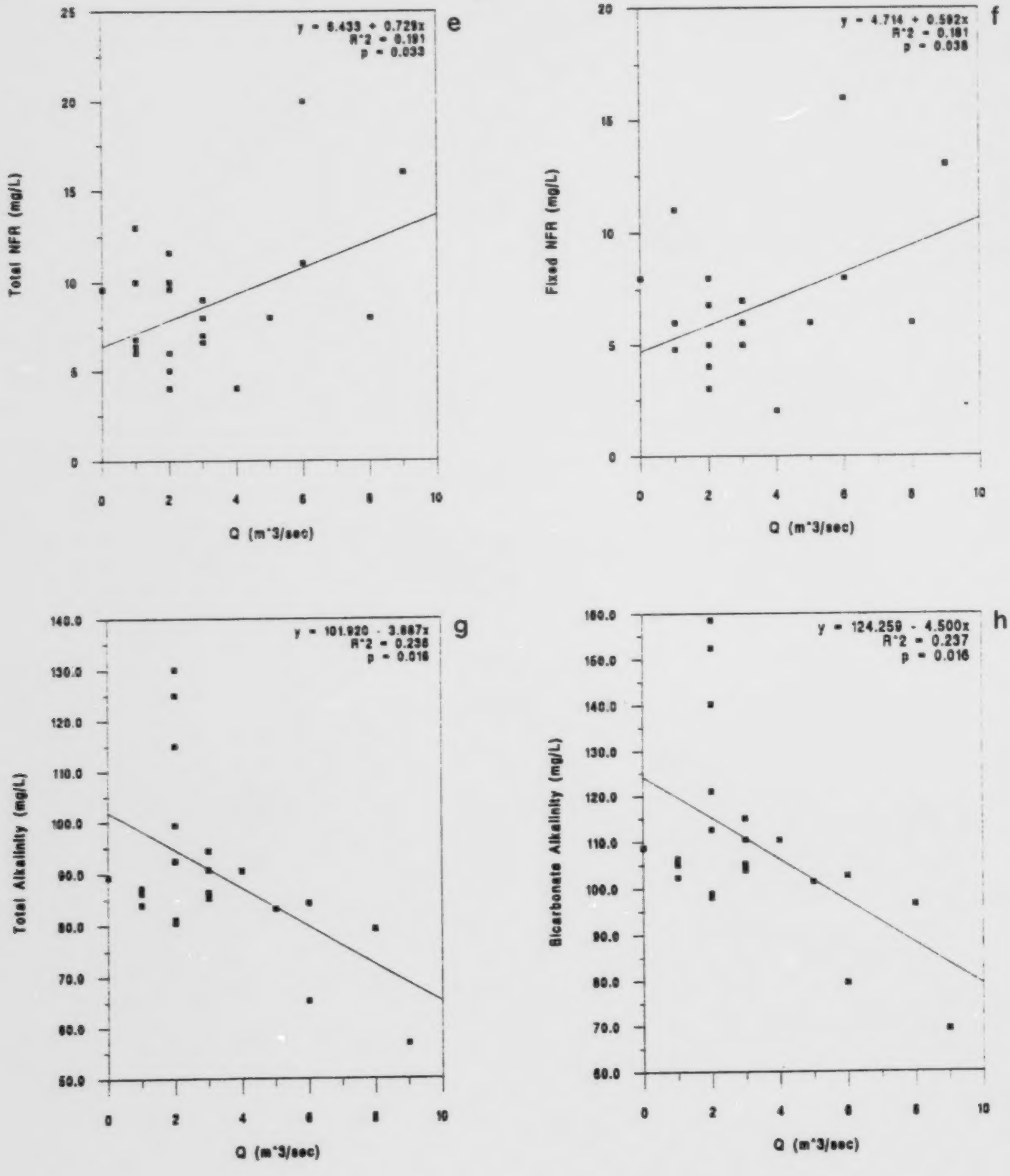
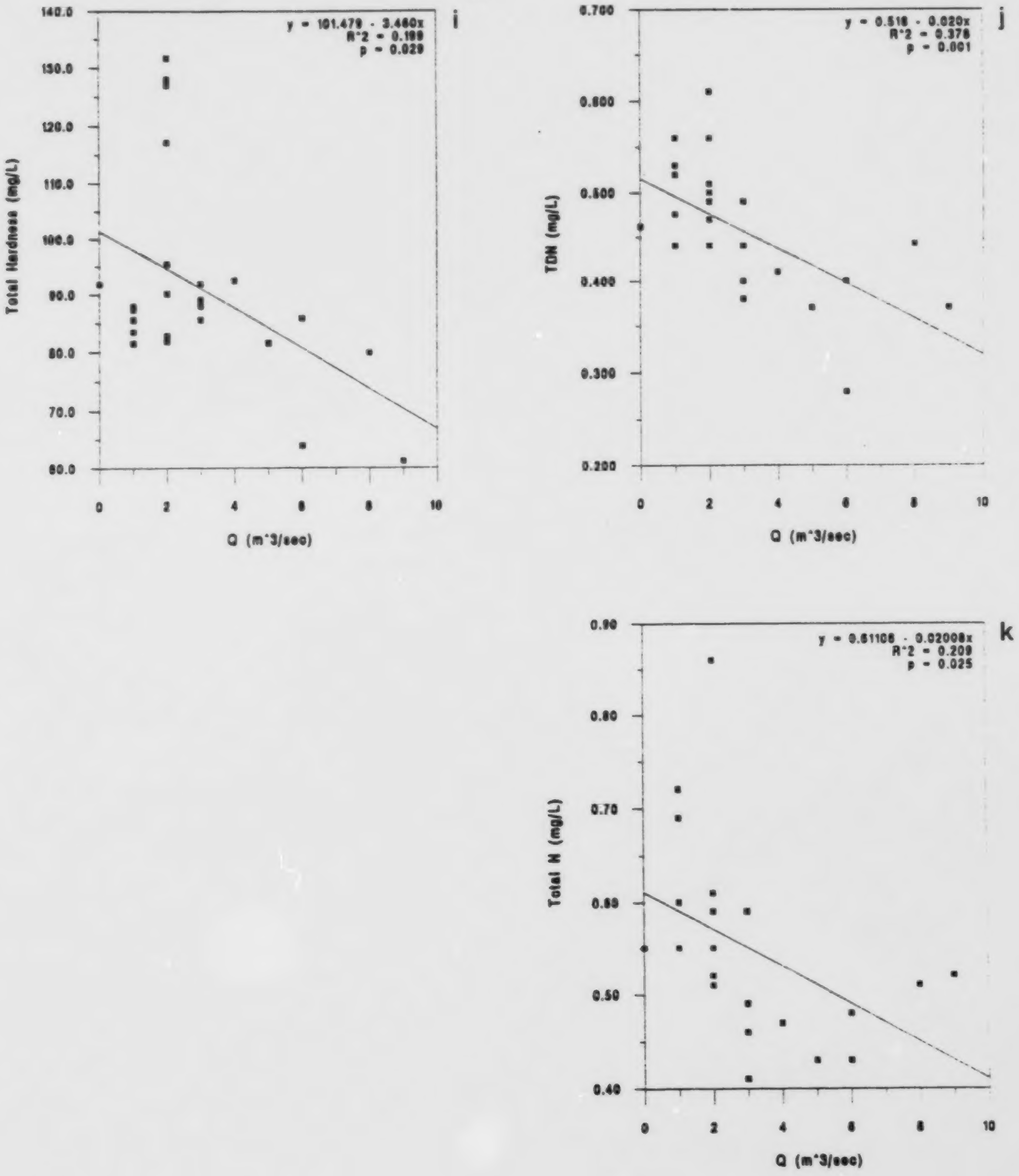


Figure 9. Continued.



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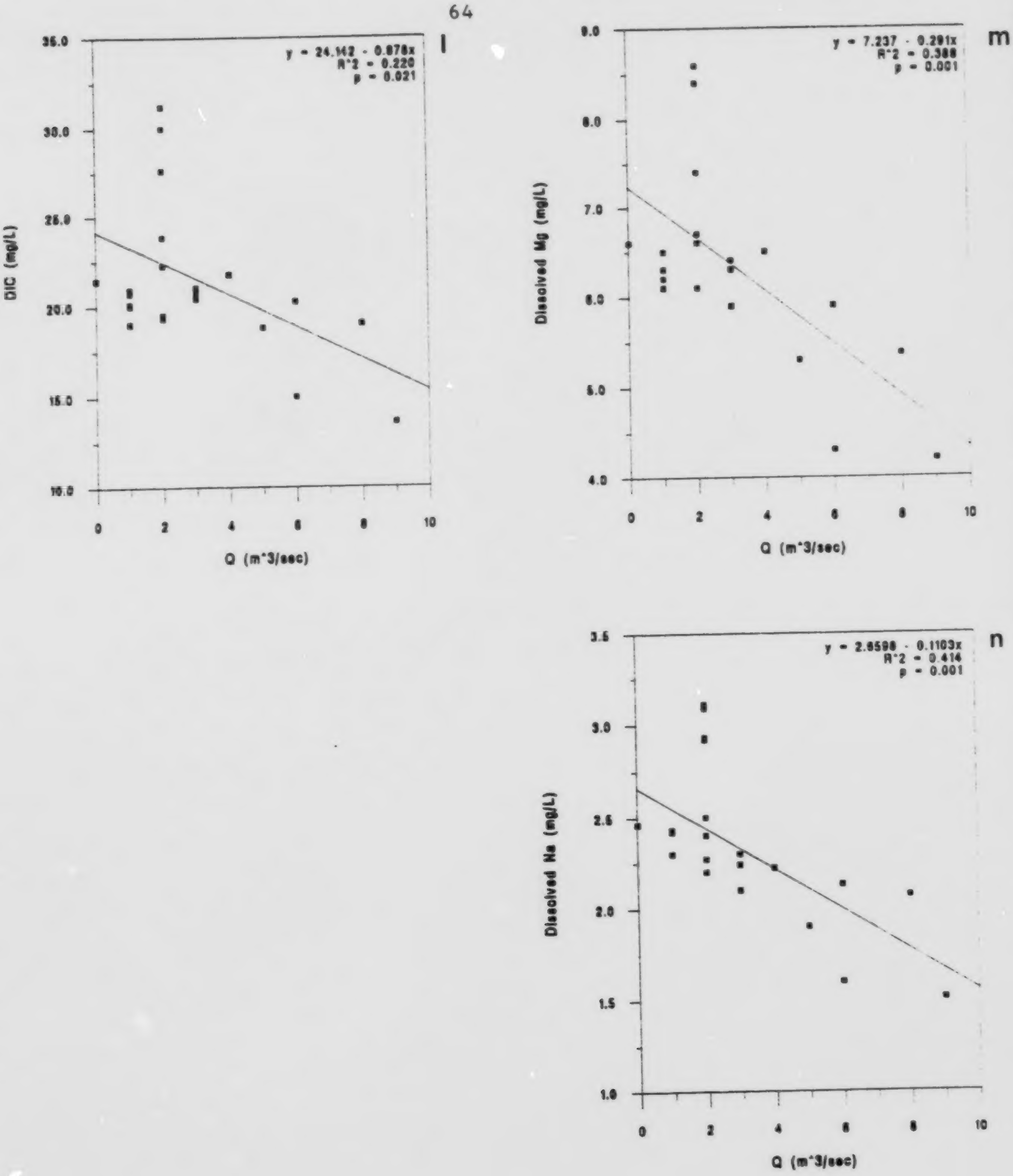

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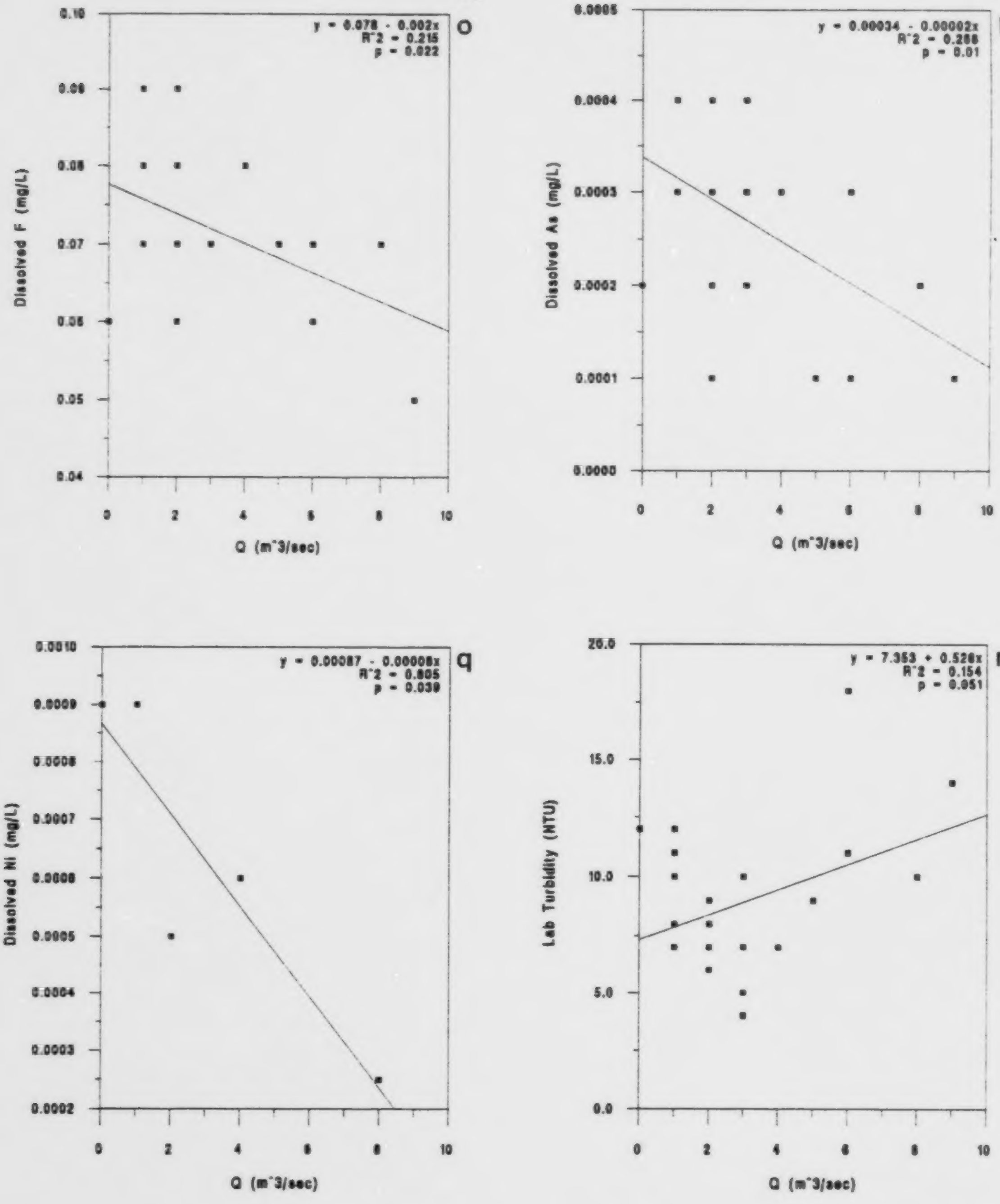

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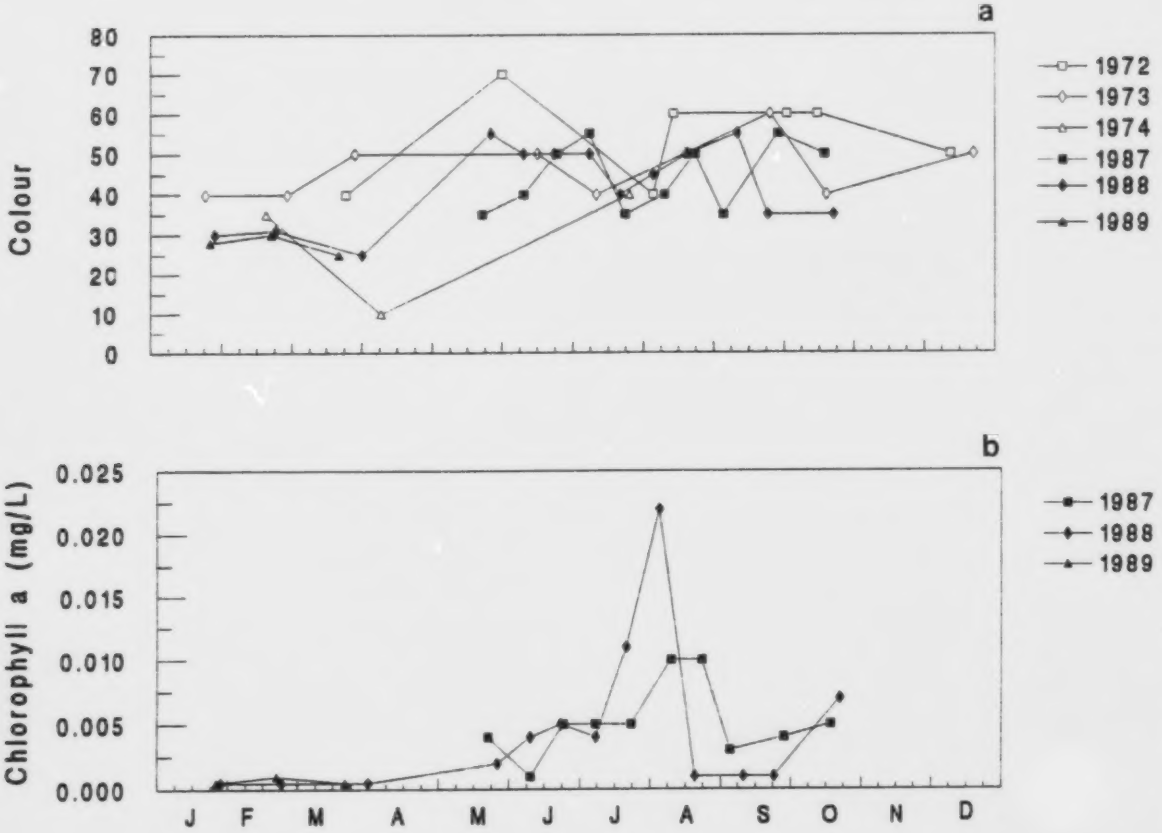


Figure 10. Seasonal water quality trends in the Burntwood River above Threepoint Lake. a. colour b. chlorophyll a c. dissolved Na d. dissolved Mg e. dissolved Zn.

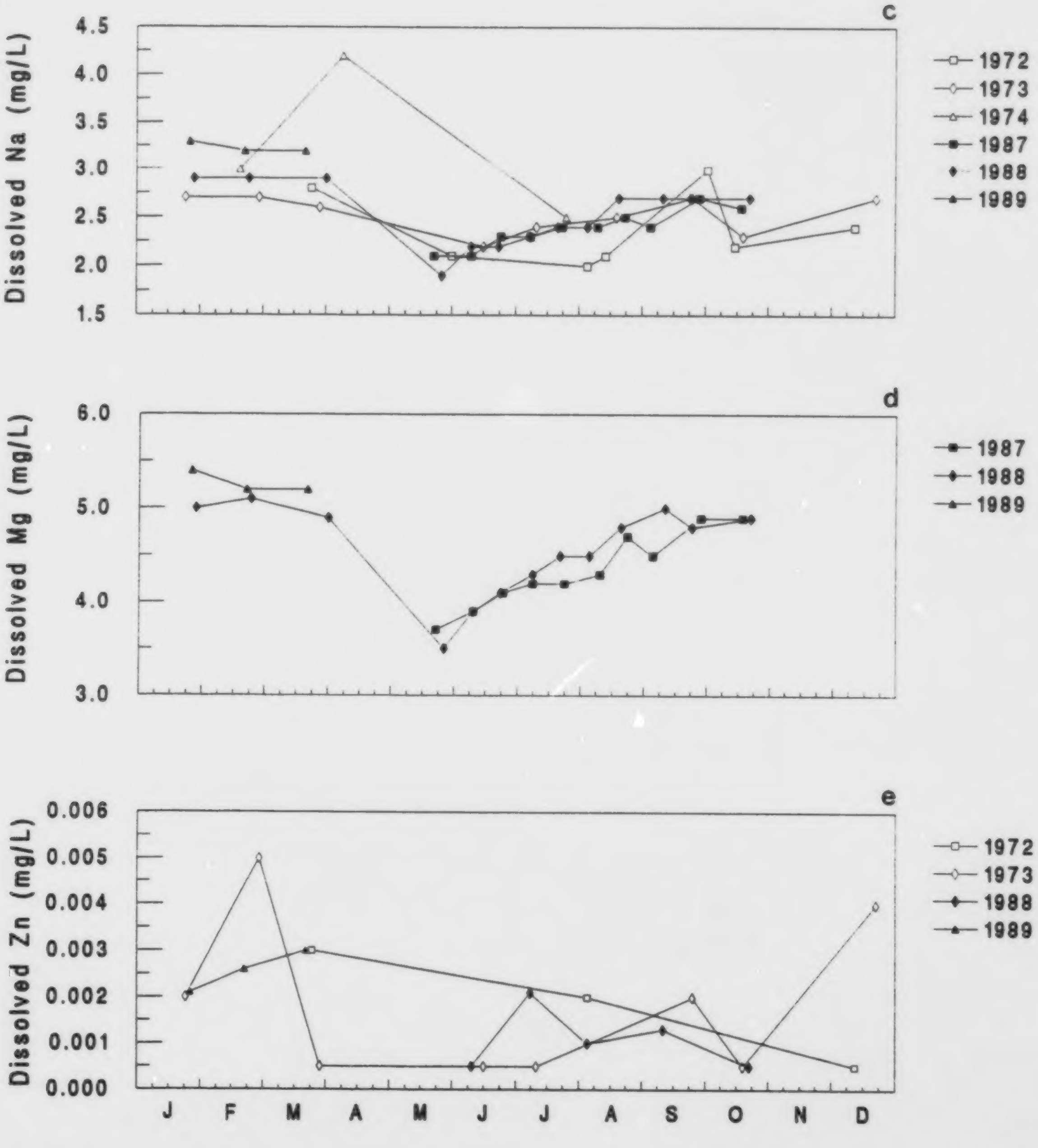


Figure 10. Continued.

season. The pre-development measurements tended to me more variable within a season, making any seasonal difference less distinct than during the FEMP study. This was the case for the pre-development Na data, in which the range during the ice-cover period overlapped that in the ice-free season although the highest values typically occurred in winter, as in the FEMP study period (Figure 10).

3.1.4.2 Footprint River above Threepoint Lake

There were clear differences in the values for many variables between the ice-free and ice-cover seasons during the 1987-1989 FEMP sampling period (Figure 11). Conductivity, alkalinity, (total and bicarbonate), hardness (total), most major ions (Ca, Mg, K, Na, Cl), reactive Si, DIC, dissolved Fe, and extractable and dissolved Mn were higher under ice cover, while dissolved Cu and chlorophyll a were lower. The lower chlorophyll a levels under ice are most probably a result of reduced light penetration through the ice and snow cover. The higher levels of some variables under ice (conductivity, hardness, alkalinity, Mg, Na, and DIC) are possibly explained by their negative correlations with river discharge and the low flows which occurred on the river in winter. Monthly mean discharges from December to April range from 1.27 to 2.49 m^3 sec^{-1} compared with 3.30 to 7.56 m^3 sec^{-1} from May to November (Water Survey of Canada 1990). This is, at best, a partial explanation as none of the other variables which increased in winter was correlated with river discharge. The lower dissolved Cu concentrations in winter also were not the product of a

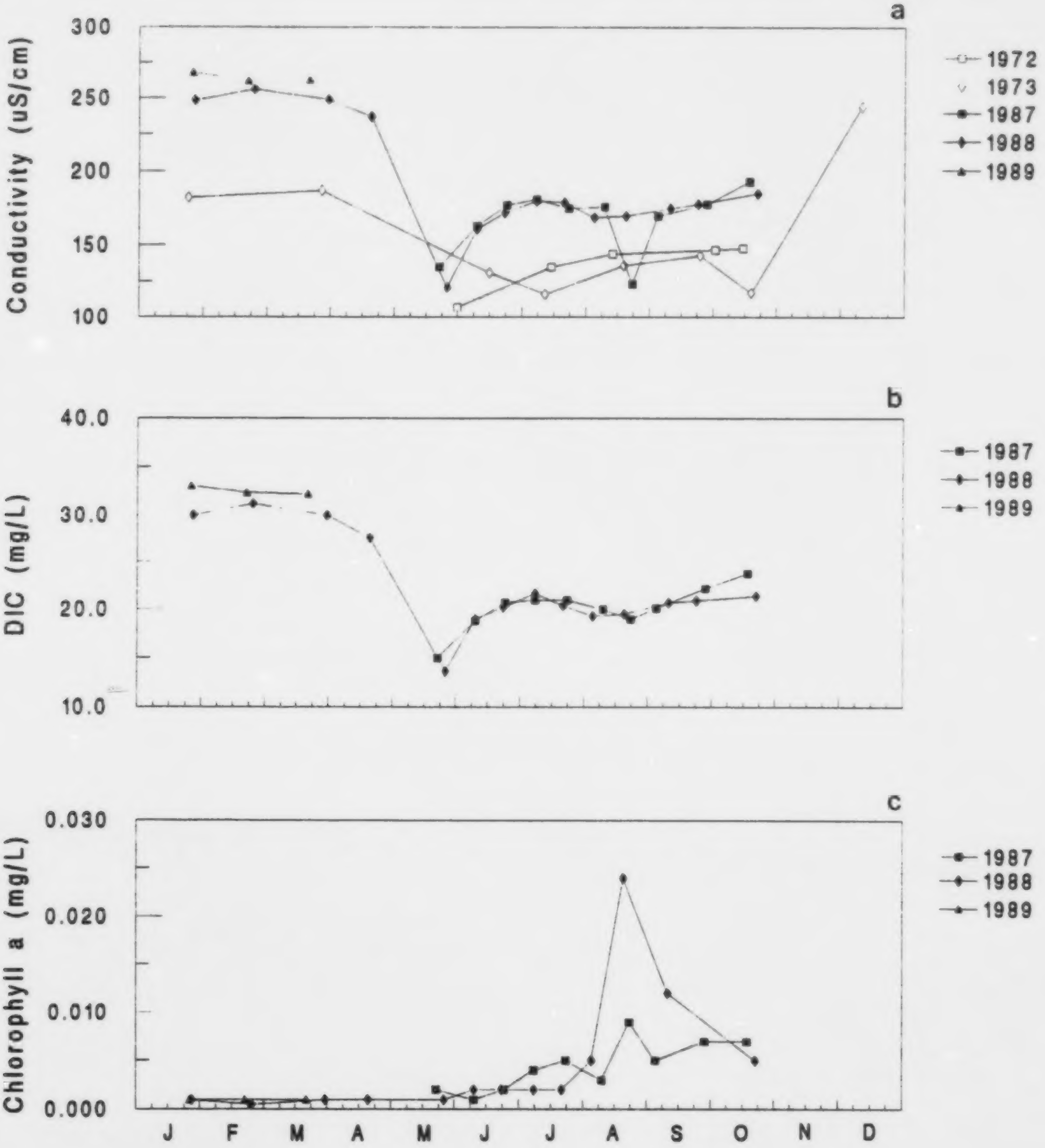
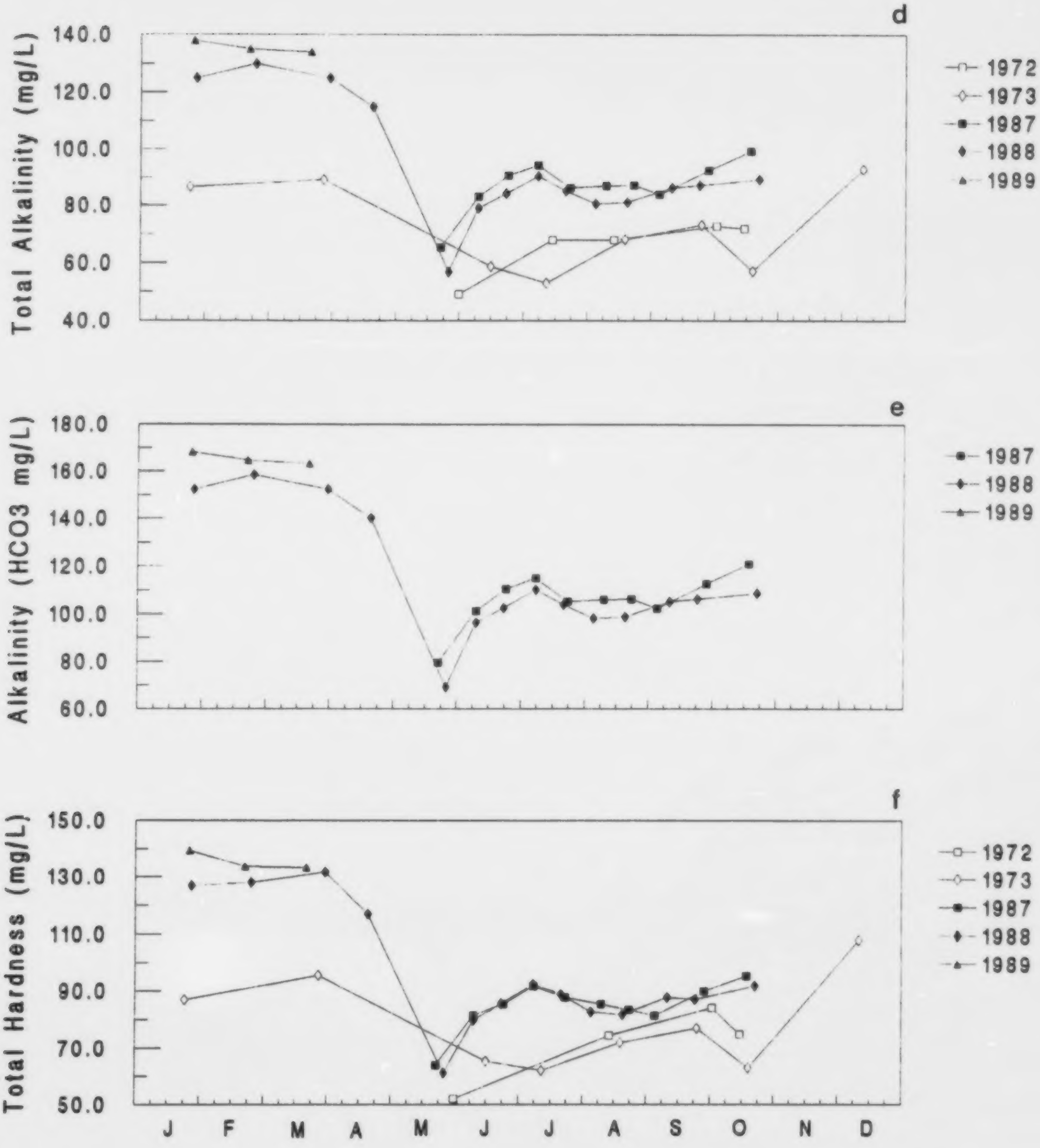


Figure 11. Seasonal water quality trends in the Footprint River above Footprint Lake. a. lab conductivity b. DIC c. chlorophyll a d. total alkalinity e. bicarbonate alkalinity f. total hardness g. dissolved Na h. reactive Si i. dissolved Cl j. dissolved Ca k. dissolved Mg l. dissolved K m. dissolved Cu n. dissolved Zn o. dissolved Fe p. extractable Mn q. dissolved Mn.



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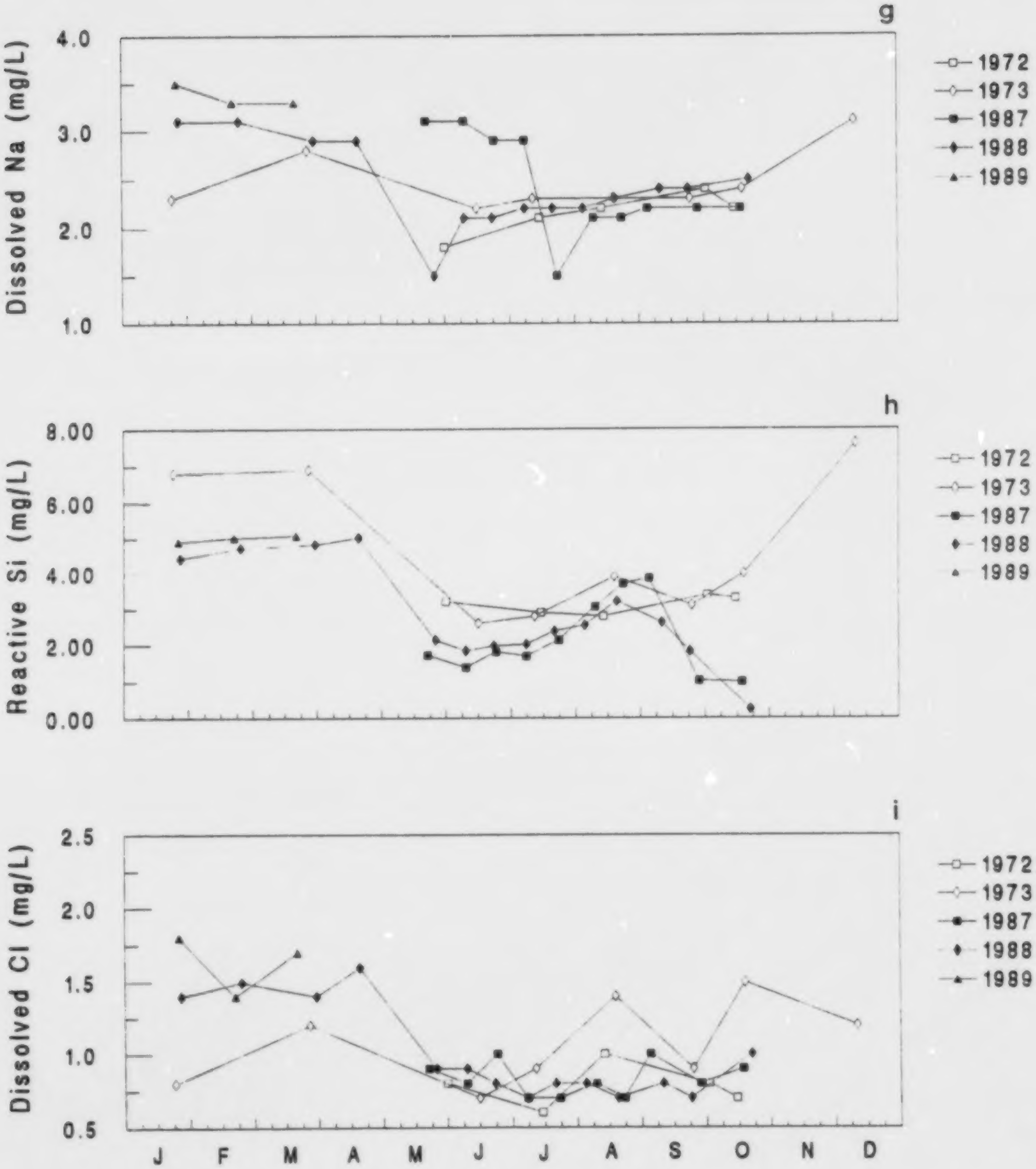

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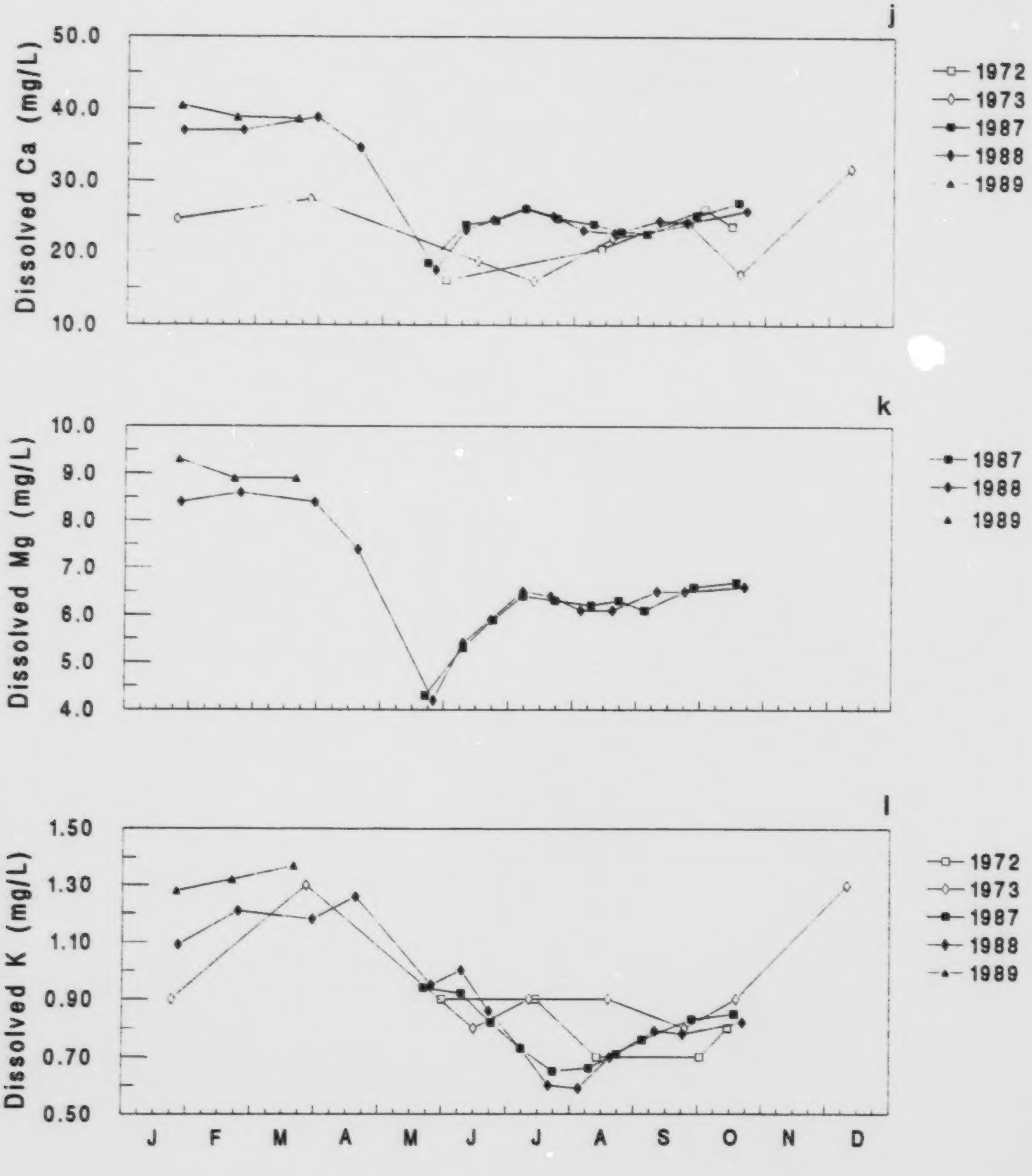

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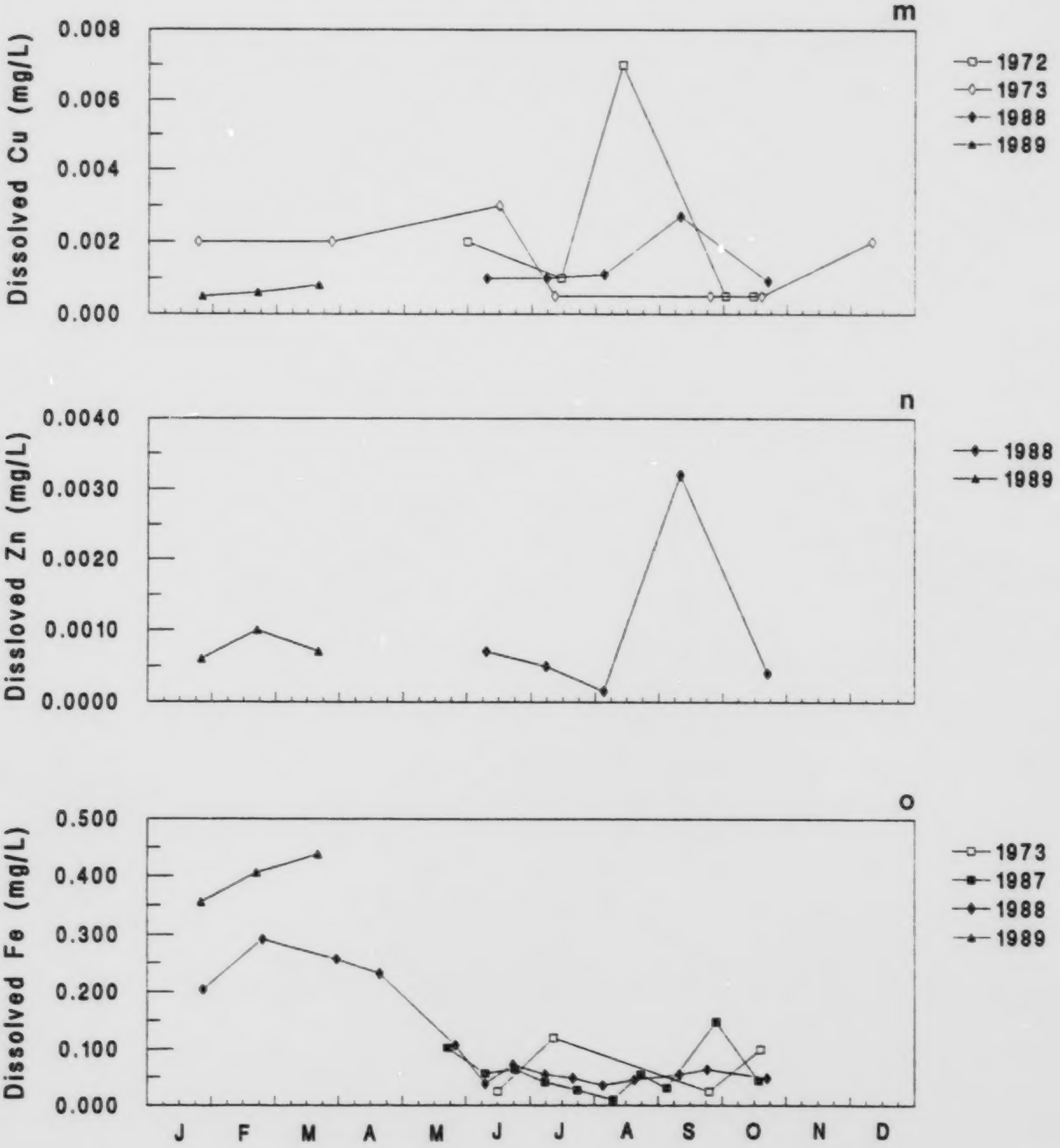

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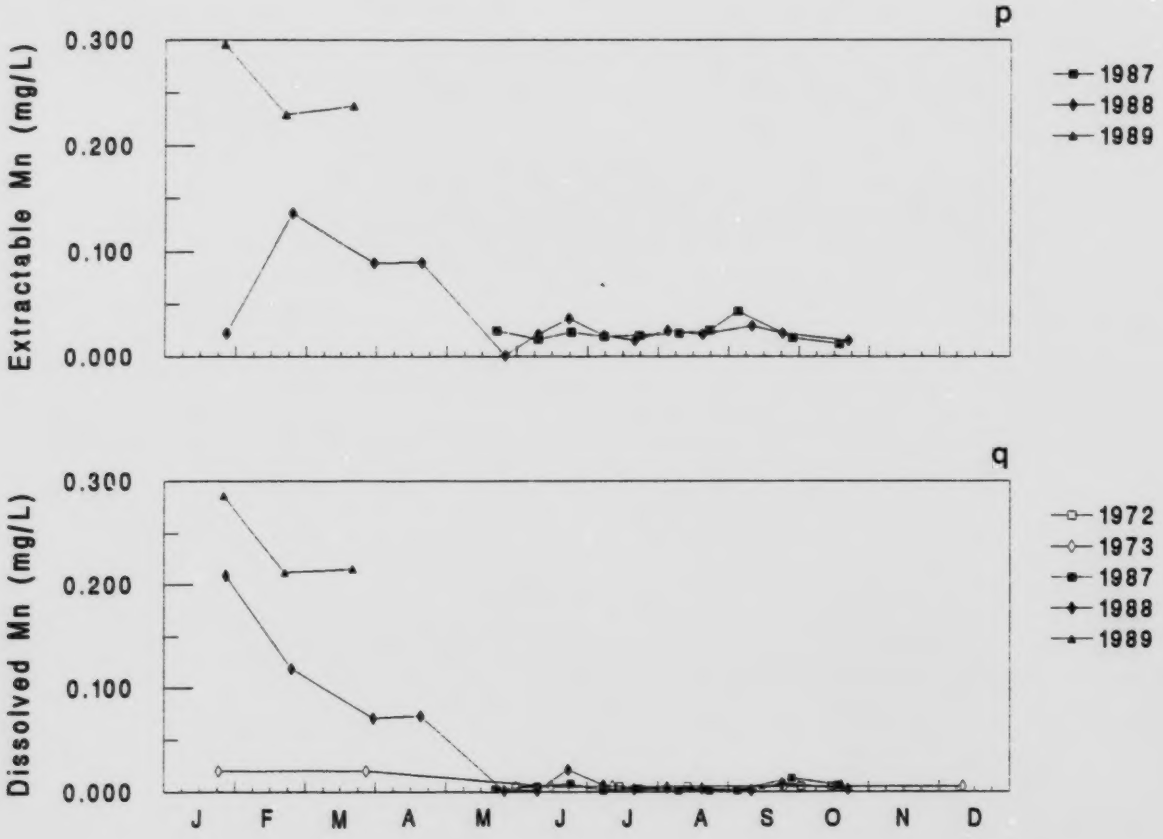


Figure 11. Continued.

relationship with discharge.

The pre-development database for this site consists of just 4 measurements of most variables under ice cover in the winters of 1972-73, and 1973-74. Qualitative examination of those variables common to the pre- and post-development periods revealed the same general pattern was present prior to development (Figure 11). Conductivity, alkalinity (total), hardness (total), ortho PO_4, dissolved Na, K, Ca, and Mn, and reactive Si all occurred in higher concentrations under ice-cover. No clear seasonal trend was evident in dissolved Cl. None of the other variables which showed distinct seasonality during the FEMP study period was measured with sufficient frequency prior to development to examine seasonal patterns. The seasonal differences during the pre-development period are probably explained by lower discharge in winter, although the nature of any water quality - water discharge relationships during the pre-development period is unknown.

3.1.4.3 Aiken River

Distinct differences between the ice-free and ice-cover seasons were evident in 9 variables (Figure 12). Alkalinity (total and bicarbonate), hardness (total), dissolved Ca, total NH_3, DIC, and dissolved As and Zn were higher under ice cover, while chlorophyll a was lower. This is generally the same pattern as in the Footprint River, suggesting a common cause. It is not known if any of these variables was correlated with discharge, however, due to the lack of flow measurements for this river.

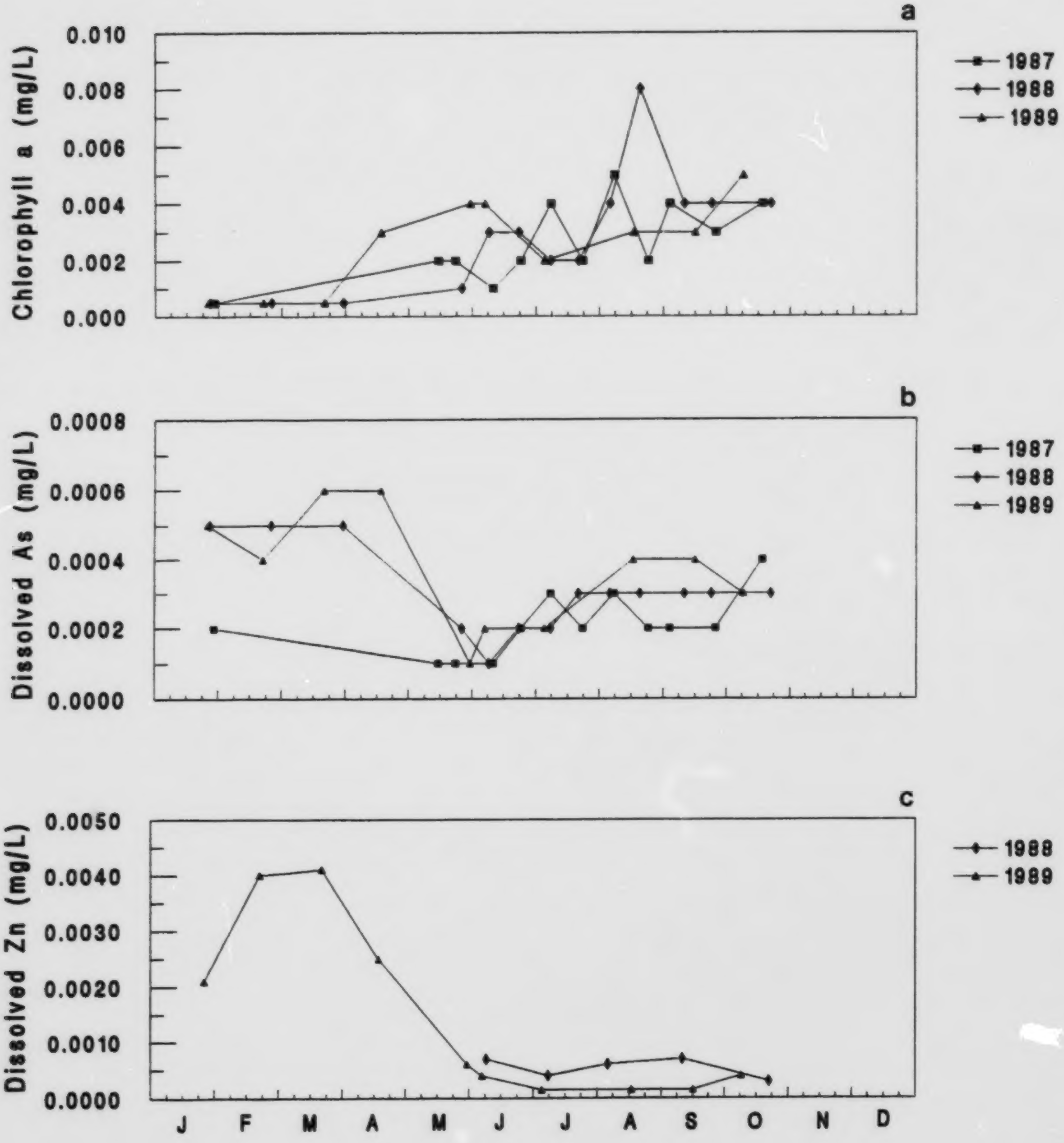
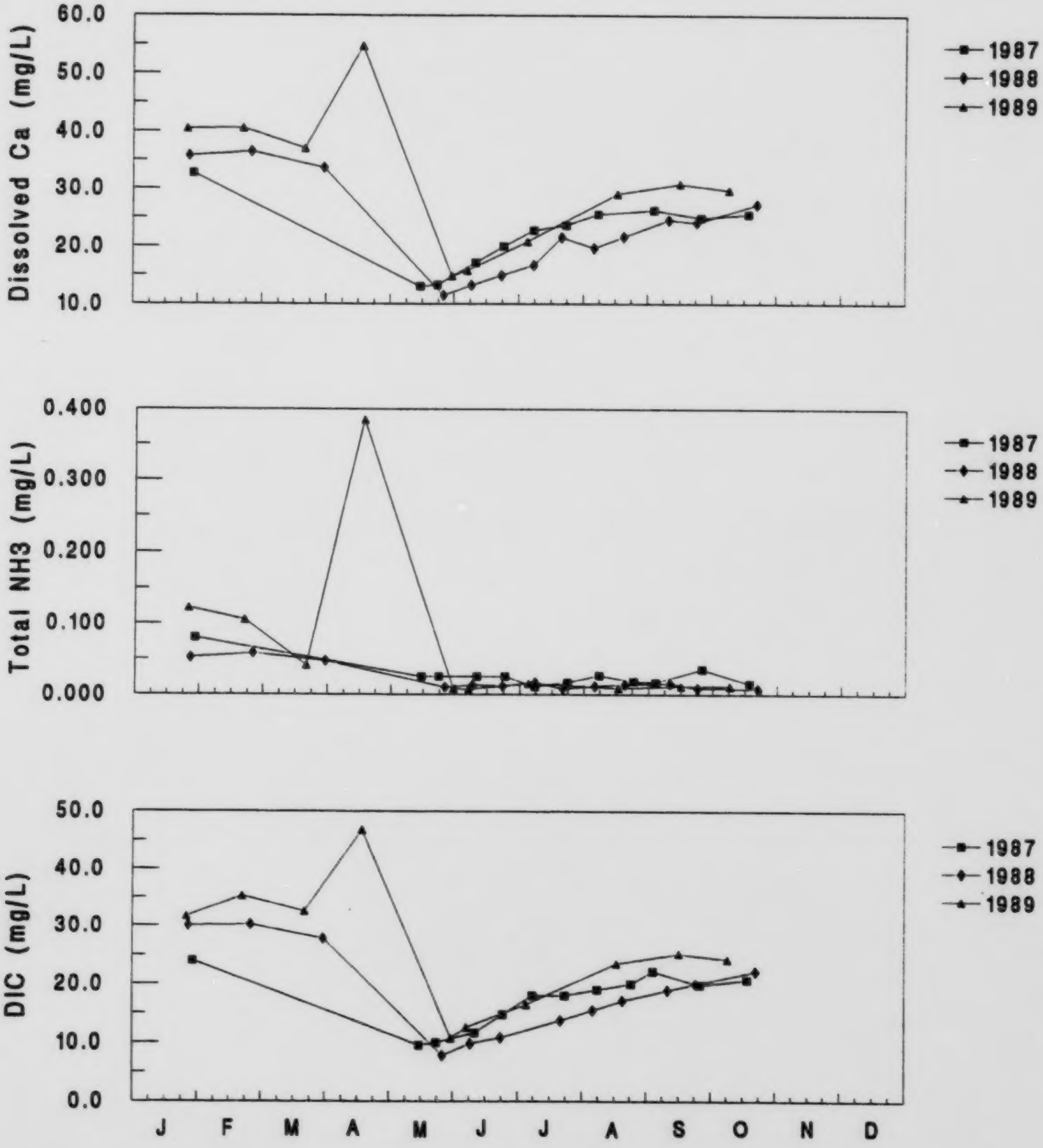


Figure 12. Seasonal water quality trends in the Aiken River. a. chlorophyll a b. dissolved As c. dissolved Zn d. dissolved Ca e. total NH_3 f. DIC g. total alkalinity h. bicarbonate alkalinity i. total hardness.



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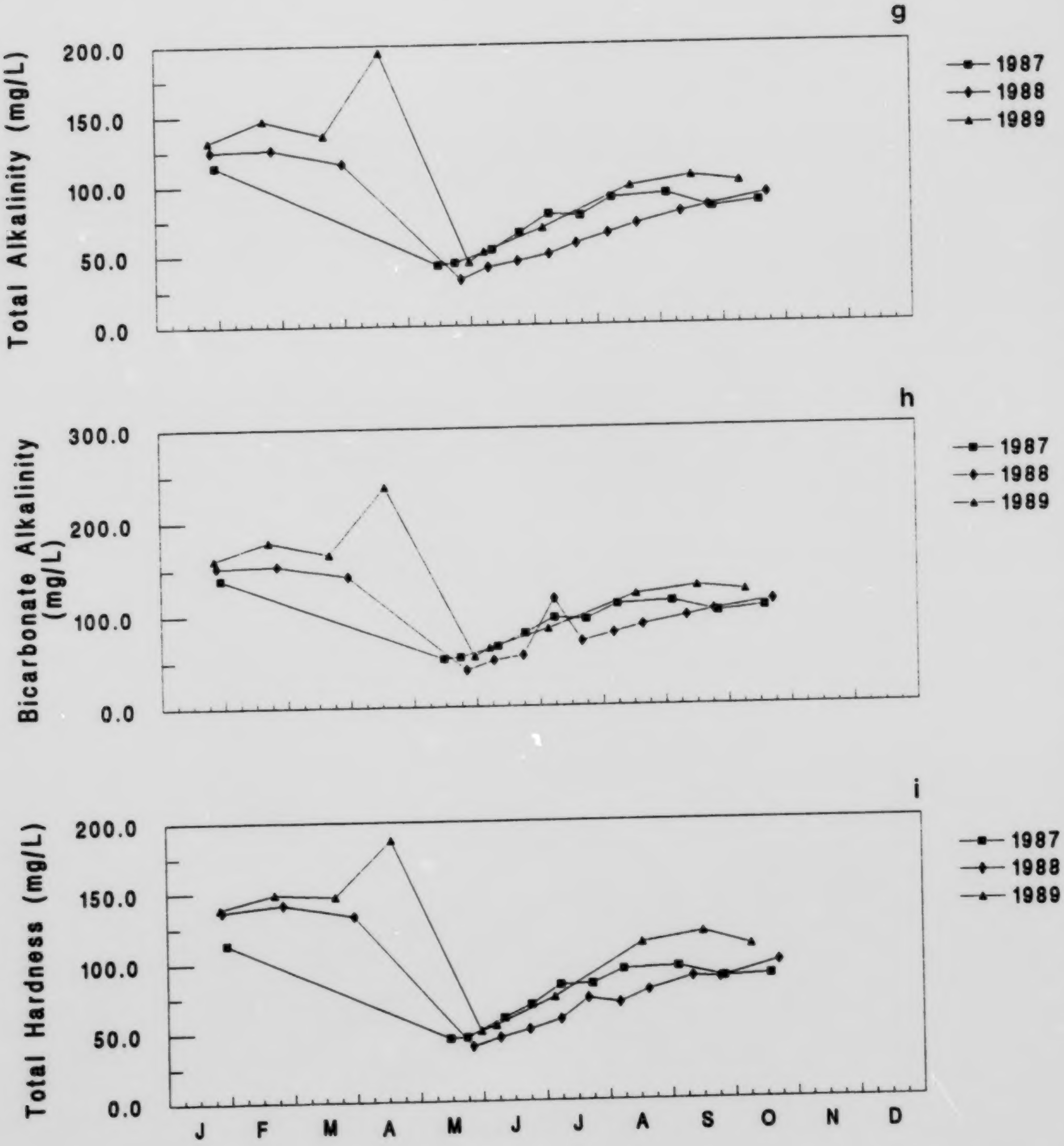


Figure 12. Continued.

3.1.5 Long-term Variation in Water Quality

Analyses of changes in water quality in the natural rivers between the pre- and post-development periods were possible only for the Burntwood and Footprint river sites. There are no pre-development water quality data for the Aiken River site.

Statistical comparisons of changes between the 1972-1973 pre-development study period and the 1987-1988 post-development period were possible for 17 variables at both sites (Tables 10 and 11). The only significant change detected in the Burntwood River was a decrease in pH, although marginally significant ($0.05<p<0.08$) decreases in dissolved Ca and SO_4 also occurred. A significant decrease in pH also was found in the Footprint River, along with decreases in NO_2-NO_3 and SO_4 and a marginally significant increase in alkalinity (total). These findings indicate a good measure of long-term stability in the chemistry of these natural rivers.

The decreases in pH at both sites probably were an artefact of undocumented modifications in analytical equipment or procedures. Equipment (e.g., probe sensitivity) and procedures (e.g., agitation for a faster but incorrect reading) for pH measurement have changed considerably in the past 15 years (M. Stainton, DFO Winnipeg, pers. comm.) although such modifications in what appears to be a simple analysis often are not considered to represent a change in method. The decreases in pH also are inconsistent with the other changes in water chemistry which occurred between the pre- and post-development study periods. An increase in acid deposition should increase the SO_4 concentration and decrease alkalinity (Schindler

Table 10. Evaluation of the significance of changes in selected water quality parameters in the Burntwood River above Threepoint Lake between 1972-1973 and 1987-1988. X_{pre} denotes the mean for the years 1972 and 1973. X_{post} denotes the mean for the years 1987 and 1988. p is the probability of Student's t-statistic for the comparison.

		pH	Conductivity µS cm^{-1}	Turbidity NTU	Tot Alk mg L^{-1}	Tot Hard mg L^{-1}	TOC mg L^{-1}	NO_2NO_3 mg L^{-1}	Ortho PO_4 mg L^{-1}	Tot P mg L^{-1}	Diss Na mg L^{-1}
Burntwood River	p	0.028	0.681	0.202	0.953	0.222	0.250	0.226	0.553	0.295	0.394
	X_{pre}	7.8	110.	14.	50.2	55.6	14.2	0.290	0.004	0.027	2.3
	X_{post}	7.43	112.[1]	19.	50.0[1]	51.1[1]	12.78[1]	0.048[1]	0.003	0.036	2.5[1]

		Diss K mg L^{-1}	Diss Ca mg L^{-1}	Diss Cl mg L^{-1}	Diss F mg L^{-1}	Diss SO_4 mg L^{-1}	React Si mg L^{-1}	Diss Fe mg L^{-1}	Diss Mn mg L^{-1}	Diss Cu mg L^{-1}	Diss Pb mg L^{-1}	Diss Zn mg L^{-1}
	p	0.859	0.062	1.00	1.00	0.066	0.095	0.240	i.d.	i.d.	i.d.	i.d.
	X_{pre}	1.0	16.3	1.0	0.07	3.3	3.6	0.10	<0.01	0.091	0.059	0.062
	X_{post}	0.94[1]	13.0[1]	1.0[1]	0.07	2.4	3.16[1]	0.125[1]	0.011[1]	0.0017	<0.0007	0.0011

i.d. Insufficient data.

[1] Discharge-standardized variables.

Table 11. Evaluation of the significance of changes in selected water quality parameters in the Footprint River above Footprint Lake between 1972-1973 and 1987-1988. X_{pre} denotes the mean for the years 1972 and 1973. X_{post} denotes the mean for the years 1987 and 1988. p is the probability of Student's t-statistic for the comparison.

		pH	Conductivity µS cm^{-1}	Turbidity NTU	Tot Alk mg L^{-1}	Tot Hard mg L^{-1}	TOC mg L^{-1}	NO_2NO_3 mg L^{-1}	Ortho PO_4 mg L^{-1}	Tot P mg L^{-1}	Diss Na mg L^{-1}
Footprint River	p	0.027	0.193	0.162	0.062	0.186	0.492	0.001	0.293	0.499	0.939
	X_{pre}	7.9	146.	7.	68.9	75.0	18.0	0.577	0.004	0.026	2.3
	X_{post}	7.50	$179.^1$	9.	90.2^1	90.1^1	11.78	0.020	0.003	0.030	2.3^1

	Diss K mg L^{-1}	Diss Ca mg L^{-1}	Diss Cl mg L^{-1}	Diss F mg L^{-1}	Diss SO_4 mg L^{-1}	React Si mg L^{-1}	Diss Fe mg L^{-1}	Diss Mn mg L^{-1}	Diss Cu mg L^{-1}	Diss Pb mg L^{-1}	Diss Zn mg L^{-1}
p	0.677	0.189	0.808	0.106	0.042	0.248	0.758	i.d.	i.d.	i.d.	i.d.
X_{pre}	0.9	22.2	1.0	0.06	4.5	3.9	0.08	<0.01	0.002	<0.001	0.003
X_{post}	0.85	25.8	0.9	0.08^1	1.8	2.49	0.084	0.021	0.0013	<0.0007	0.0010

i.d. Insufficient data.

[1] Discharge-standardized variables.

et al. 1980), yet decreases in SO_4 appear to have occurred at both sites and an increase in alkalinity occurred in the Footprint River.

There is no obvious explanation for the 25-60% decreases in SO_4 at both sites nor for the almost 96% decrease in NO_2-NO_3 in the Footprint River. The same declining trend in NO_2-NO_3 was evident in the Burntwood River as well, although the 83% lower value during the FEMP study was not significantly different from the pre-development value (Table 10). The higher mean concentrations of both parameters during the pre-development period were not the product of outliers or occasional high values, concentrations of both variables were consistently higher throughout the pre-development study period.

It is particularly difficult to understand how the much lower NO_2-NO_3 concentrations during the FEMP study could be a product of natural variation. Measurements of NO_3 concentrations reported in a pre-development study of the Rat-Burntwood system suggest problems with the analytical method used to produce the NO_2-NO_3 data in the NAQUADAT file may be the cause. Concentrations of NO_3 in the Rat-Burntwood system, including Threepoint Lake, reported by Cleugh (1974) were in the range of <0.001 to 0.007 mg L^{-1} between June and September, with values in the range of 0.050 to 0.125 mg L^{-1} under ice cover. Concentrations of NO_2-NO_3 measured during the FEMP study were comparable to the NO_3 concentrations reported by Cleugh. Although Cleugh just measured NO_3, this is the dominant contributor to NO_2-NO_3 in the Rat-Burntwood system. Concentrations of NO_2 in the system typically are <0.001 mg L^{-1} between June and September (Berger and Ramsey 1991). Long-term monitoring of NO_3 levels in natural waters at the Experimental Lakes Area in northwestern Ontario, using the same method employed by Cleugh, have not shown the dramatic decreases evident in the NAQUADAT data. The trend is for slightly increasing NO_3 concentrations due to increasing atmospheric deposition (R.E. Hecky, DFO, Winnipeg, MB. pers. comm.). Because of the uncertain validity of the pre-development NO_2-NO_3 data in the NAQUADAT file, any decreases detected in the analysis of temporal changes associated with hydroelectric development which were based on NAQUADAT data were considered to be an artefact unless they were supported by measurements from other sources.

There appears to be a similar problem with the pre-development SO_4 measurements. Decreases of 31 to 60% seem to have occurred in the Burntwood and Footprint rivers and in the Churchill River near Leaf Rapids (pre: 3.6 mg L^{-1}, NAQUADAT; post: 2.5 mg L^{-1}, G. McCullough, DFO, Winnipeg, MB. unpubl. data.), all natural sites, between the pre-development study period and the FEMP study. Again it seems unlikely that such a change might naturally occur in 3 separate drainage basins. A change in SO_x emissions from the INCO smelter in Thompson also can be ruled out as all 3 sites lie upwind of this source. Again, the most likely cause of this shift is a change in analytical methods or laboratories. Because of this uncertainty in the pre-development SO_4 data, it generally was not possible to determine if the hydroelectric developments had any impact on SO_4 concentrations.

The pre-development database unfortunately is deficient in its coverage of metal species. In particular, measurements of extractable or total metals generally were made on fewer than 4 dates between 1971 and 1974. Several dissolved metal species were routinely analyzed (Tables 10 and 11), although occasionally with higher limits of detection than employed in the FEMP study. The FEMP study also contributed to this poor coverage of metals with its limited examination of dissolved species. None of dissolved Mn, Cu, Pb, or Zn was analyzed in 1987, and sampling was terminated after March 1989, resulting in just one full calendar year of study. This eliminated the possibility of a statistical analysis for these species. Nevertheless, qualitative comparison of the available data suggests dissolved Cu, Pb, and Zn levels perhaps were lower during the FEMP study, while dissolved Mn may have been higher (Tables 10 and 11). There was no significant change in the dissolved Fe concentration.

3.1.6 Uses of the Natural River Data

The natural rivers were originally included in the FEMP study in the hope that they might serve as surrogates for pre-development conditions at sites where historical data were deficient or lacking (Seidl 1987). However, there is no means of establishing the validity for their general use as control sites. This would require a long-term comparison of water quality in the regulated and natural rivers prior to development to demonstrate similarities in water quality and in their responses to variations in climatic conditions. This sort of comparison was not possible for any of the natural rivers included in this study because of the limited pre-development databases for all of the sites. Data from the site on the Burntwood River above Threepoint Lake clearly are useful for interpretation of the cause and nature of water quality changes in the lower Burntwood River below Threepoint Lake, as this was the dominant source of flow prior to diversion. It is quite unlikely that current water quality in the natural rivers would be quantitatively comparable to pre-development water quality in the Nelson or Churchill rivers due to the large differences in discharge and drainage area. The natural rivers would be dominated by the influence of local geology and soils, while the larger Churchill and Nelson integrate the effects of geology, soils, land use, and climate from as far west as the Rocky Mountains. Therefore, the natural river data were not generally employed as controls in the analysis of water quality changes due to hydroelectric development although the Burntwood and Footprint river data were used to assist in identification of the cause of water quality changes on the Rat and Burntwood rivers. The natural river data also are useful for interpretation of chemical budgets in the Churchill-Nelson system as they can be used to define the limits of natural runoff contributions to chemical fluxes in the regulated system.

3.2 Norway House Area

3.2.1 Nelson River below Sea River Falls

3.2.1.1 Current Water Quality

With a few important exceptions, physical and chemical water quality at this site generally was well within Canadian guidelines for human consumption, recreational use, and the protection of aquatic life. The exceptions included turbidity, which always exceeded the drinking water guideline, and extractable Al and total Cu, for which the mean concentrations exceeded the respective guidelines for protection of aquatic life by factors of 1.8 and 2.4 (Table 12). Other variables (extractable Fe and Mn, colour) occasionally exceeded the drinking water guideline. These sporadic exceedences are characteristic of natural rivers and streams in northern Manitoba, occurring at all of the sites sampled in this study.

3.2.1.2 Water Quality - Water Discharge Relationships

A number of variables were negatively correlated with Nelson River discharge during the 1987-1989 study period (Figure 13). These included pH (lab), alkalinity (total and bicarbonate), hardness (total and non-carbonate), conductivity (field and lab), TDS, most major ions (Ca, Mg, K, Na, Cl, SO_4), dissolved NO_2-NO_3, total NH_3 and DIC. Positive correlates with discharge included turbidity (field) and NFR (total and fixed), particulate N, POC, and extractable Fe. The sparse pre-development water quality data precluded an analysis of changes in discharge relationships since

Table 12. Summary statistics for physical and chemical water quality variables measured in the Nelson River below Sea River Falls; January, 1987, to September, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.8	0.3	8.3	7.3
	Lab		7.86	0.20	8.13	7.49
Conductivity	Field	$\mu S\ cm^{-1}$	292	30	331	220
	Lab		309	36	355	212
Turbidity	Field	NTU	10^{+}	3	16^{+}	6^{+}
	Lab		8^{+}	3	13^{+}	4
Colour	True	Relative	9	4	20^{+}	3
TDS		$mg\ L^{-1}$	163	19	189	124
NFR	Total	$mg\ L^{-1}$	6	3	10	3
	Fixed	$mg\ L^{-1}$	5	3	10	2
Alkalinity	Total	$mg\ L^{-1}$	103.0	10.6	120.0	82.3
	Bicarbonate	$mg\ L^{-1}$	125.6	12.9	146.3	100.3
Hardness	Total	$mg\ L^{-1}$	117.7	12.9	138.7	89.4
	Non-carbonate	$mg\ L^{-1}$	14.7	3.6	20.9	4.5
Free CO_2		$mg\ L^{-1}$	3.53	2.53	8.62	0
Phosphorus	Particulate	$mg\ L^{-1}$	0.013	0.007	0.026	<0.006
	Ortho PO_4	$mg\ L^{-1}$	<0.003	0.002	0.010	<0.003
	Total Dissolved	$mg\ L^{-1}$	0.010	0.005	0.023	0.004
	Total	$mg\ L^{-1}$	0.023	0.006	0.032	0.010
Nitrogen	Particulate	$mg\ L^{-1}$	0.051	0.022	0.090	<0.010
	Total Dissolved	$mg\ L^{-1}$	0.300	0.034	0.370	0.230
	NO_2-NO_3	$mg\ L^{-1}$	0.013	0.014	0.047	<0.010
	Total NH_3	$mg\ L^{-1}$	0.023	0.012	0.050	0.006
	Total	$mg\ L^{-1}$	0.35	0.04	0.43	0.27
Carbon	DOC	$mg\ L^{-1}$	7.63	0.67	8.98	6.55
	POC	$mg\ L^{-1}$	0.385	0.145	0.760	0.160
	TOC	$mg\ L^{-1}$	8.02	0.63	9.56	7.02
	DIC	$mg\ L^{-1}$	25.0	2.2	28.8	19.0
Chlorophyll-a		$mg\ L^{-1}$	0.002	0.002	0.006	<0.001
Ca	Dissolved	$mg\ L^{-1}$	29.3	3.1	34.6	22.3
Mg	Dissolved	$mg\ L^{-1}$	10.8	1.2	12.7	8.2
K	Dissolved	$mg\ L^{-1}$	2.27	0.25	2.67	1.84
Na	Dissolved	$mg\ L^{-1}$	15.9	2.5	19.6	10.9

Table 12. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	0.91	0.54	2.40	0.25
Cl	Dissolved	mg L^{-1}	18.2	3.4	23.8	12.0
F	Dissolved	mg L^{-1}	0.10	0.01	0.14	0.08
SO_4	Dissolved	mg L^{-1}	24.1	3.7	28.4	16.3
Al	Extractable	mg L^{-1}	0.181^{+++}	0.094	0.385^{+++}	<0.100
	Dissolved	mg L^{-1}	<0.100	0.015	0.100	<0.100
As	Dissolved	mg L^{-1}	0.0006	0.0001	0.0008	0.0005
Ba	Total	mg L^{-1}	<0.08	0.01	0.10	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L^{-1}	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0001	0.0007	<0.0005
Cu	Total	mg L^{-1}	0.0047^{+++}	0.0054	0.0208^{+++}	0.0011
	Dissolved	mg L^{-1}	0.0019	0.0006	0.0027	0.0011
Fe	Extractable	mg L^{-1}	0.197	0.072	0.365$^{+,+++}$	0.087
	Dissolved	mg L^{-1}	0.033	0.016	0.063	0.010
Pb	Total	mg L^{-1}	0.0008	0.0005	0.0020	<0.0007
	Dissolved	mg L^{-1}	0.0011	0.0020	0.0073	<0.0007
Mn	Extractable	mg L^{-1}	0.016	0.012	0.067^{+}	0.005
	Dissolved	mg L^{-1}	0.004	0.003	0.011	<0.002
Ni	Total	mg L^{-1}	0.0012	0.0004	0.0022	<0.0005
	Dissolved	mg L^{-1}	0.0009	0.0003	0.0016	0.0007
Se	Dissolved	mg L^{-1}	0.0002	0.0001	0.0003	<0.0001
V	Total	mg L^{-1}	0.0007	0.0005	0.0015	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0002	0.0007	<0.0005
Zn	Total	mg L^{-1}	0.0032	0.0022	0.0083	0.0009
	Dissolved	mg L^{-1}	0.0009	0.0007	0.0024	<0.0003
Discharge	Daily	m^3 sec^{-1}	291	163	521	0

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recreational use.

$^{+++}$ Exceeds limit for protection of aquatic life.

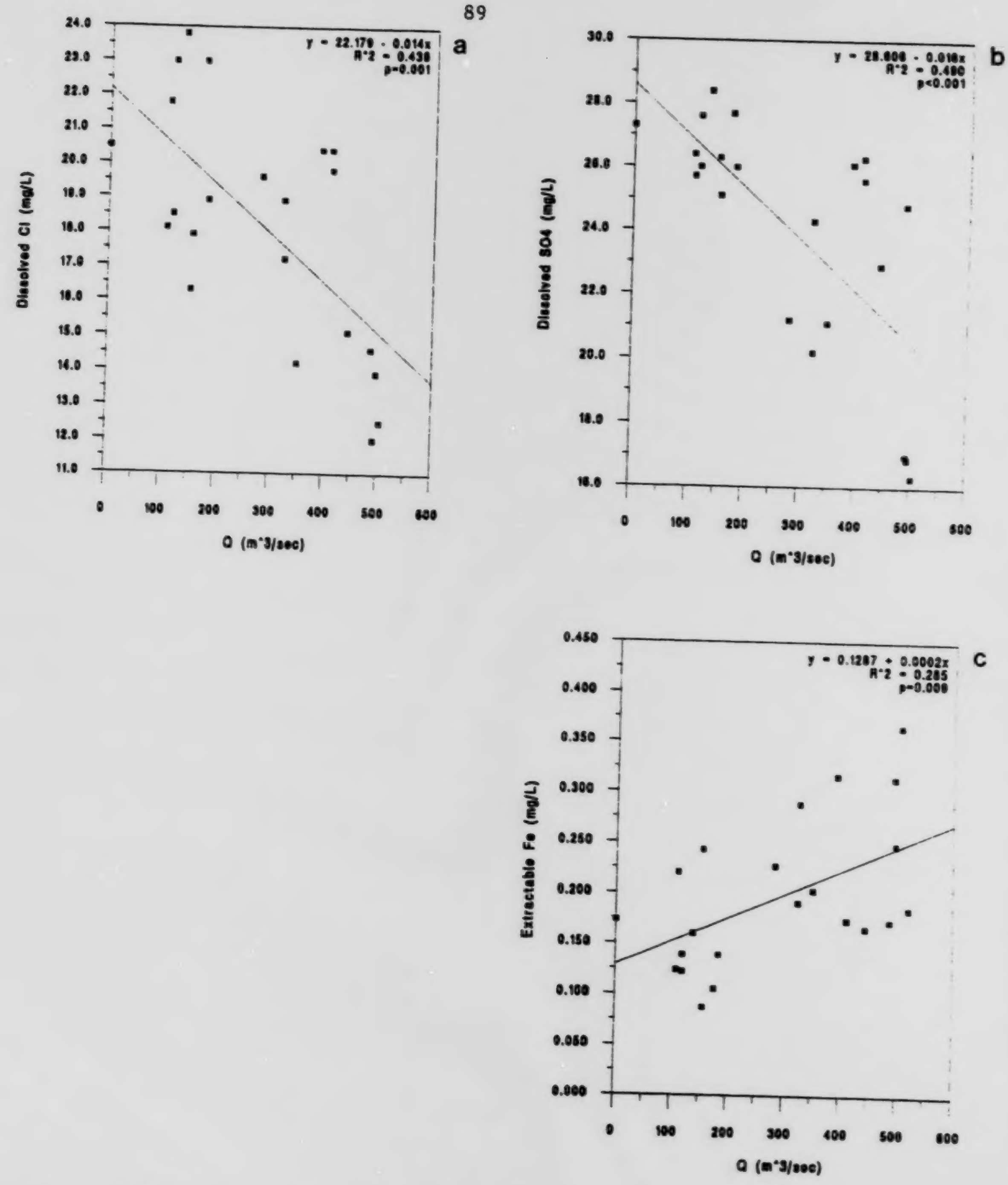
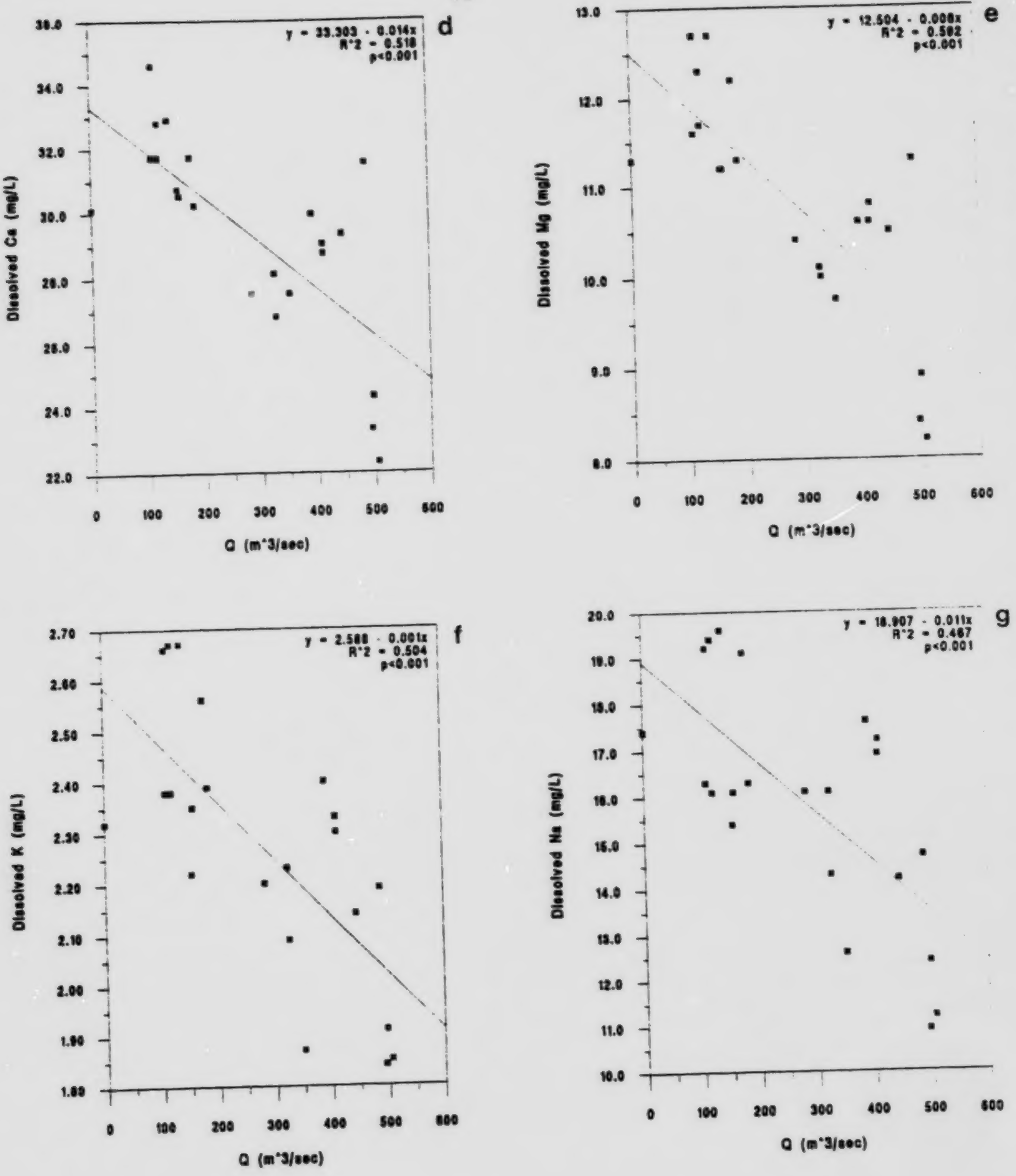


Figure 13. Water chemistry - water discharge relationships in the Nelson River below Sea River Falls, 1987-1989. a. dissolved Cl b. dissolved SO_4 c. extractable Fe d. dissolved Ca e. dissolved Mg f. dissolved K g. dissolved Na h. dissolved NO_2-NO_3 i. total NH_3 j. POC k. DIC l. bicarbonate alkalinity m. particulate N n. total hardness o. non-carbonate hardness p. TDS q. total alkalinity r. total NFR s. fixed NFR t. lab pH u. field turbidity v. field conductivity w. lab conductivity.



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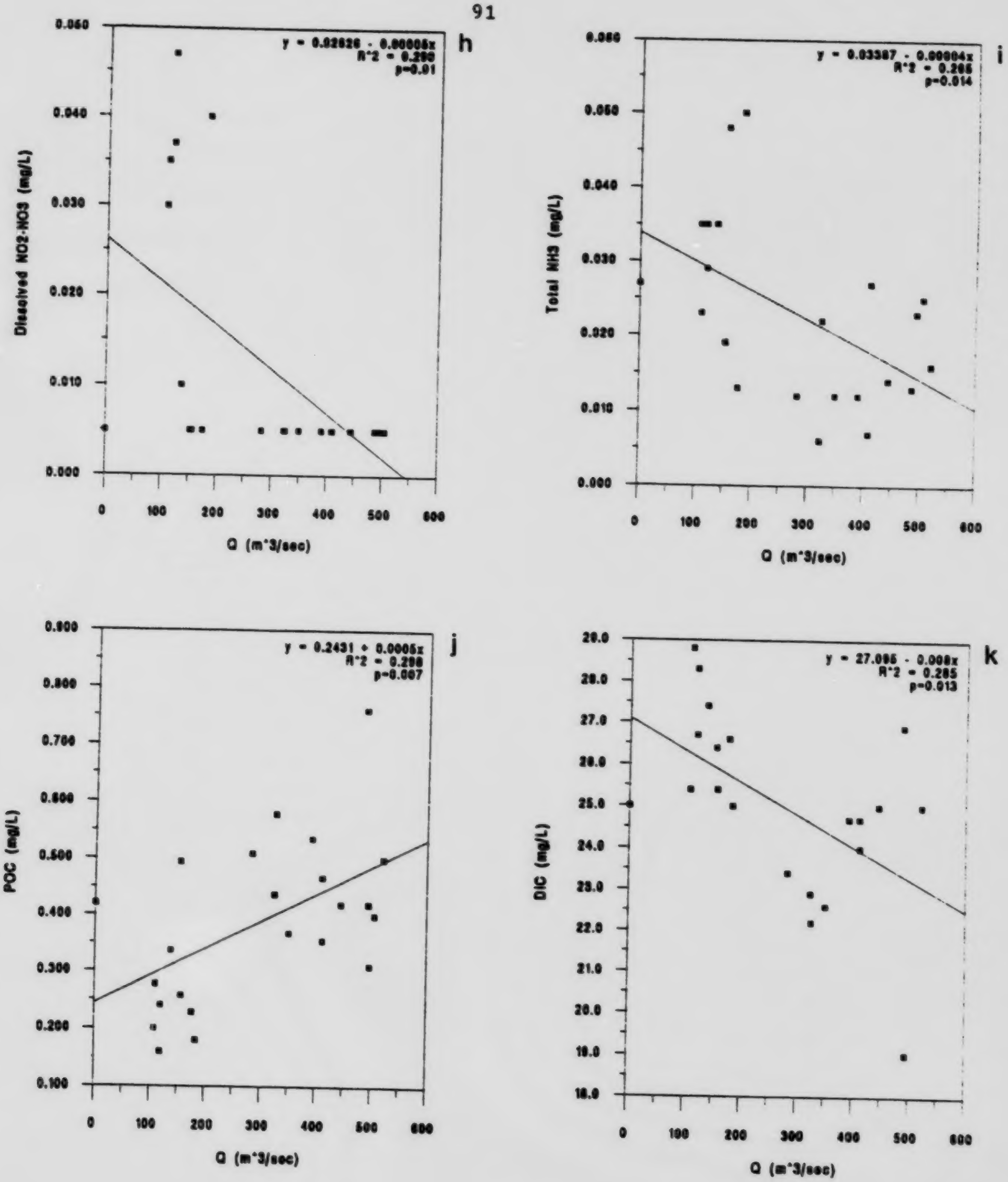

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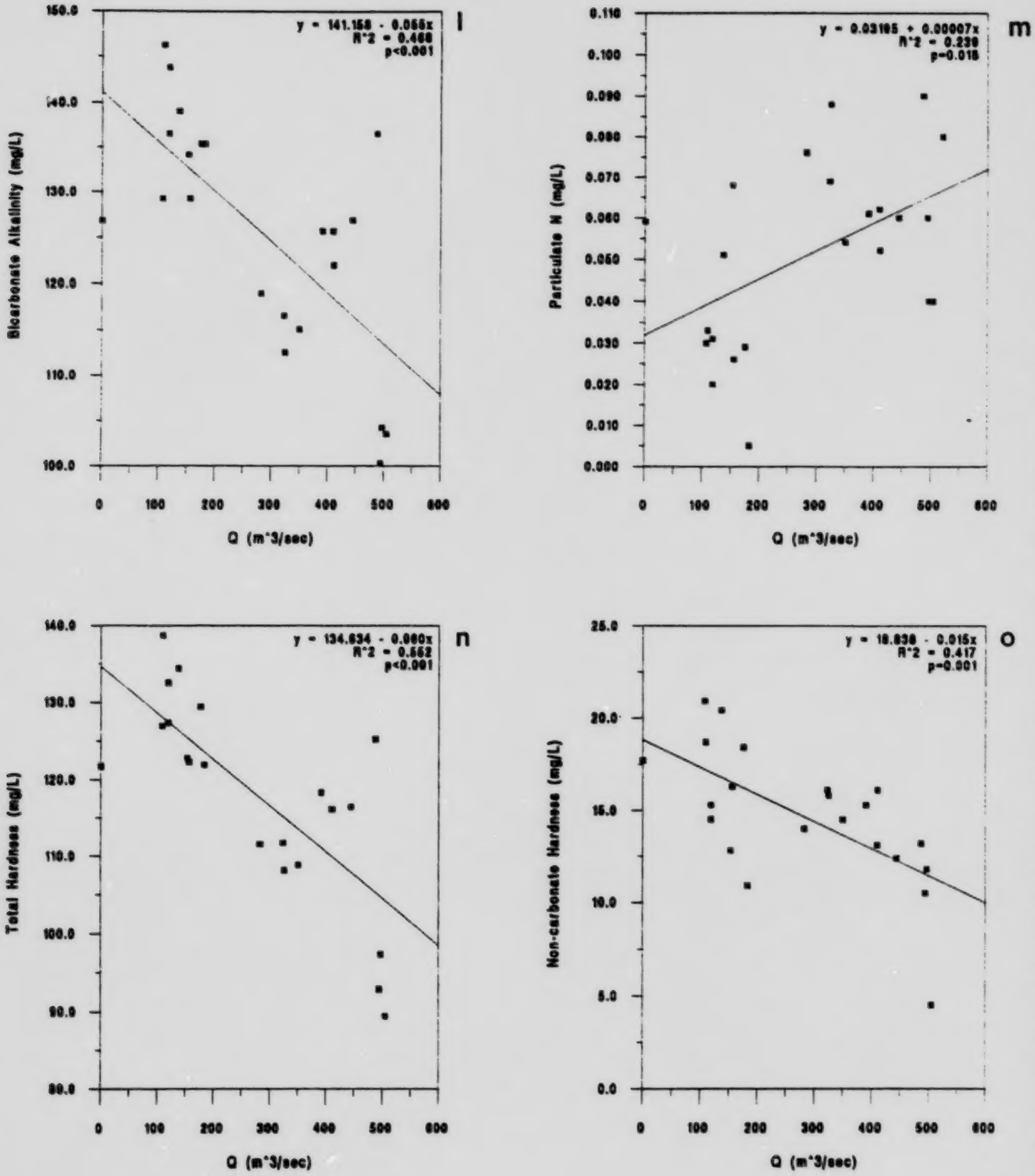

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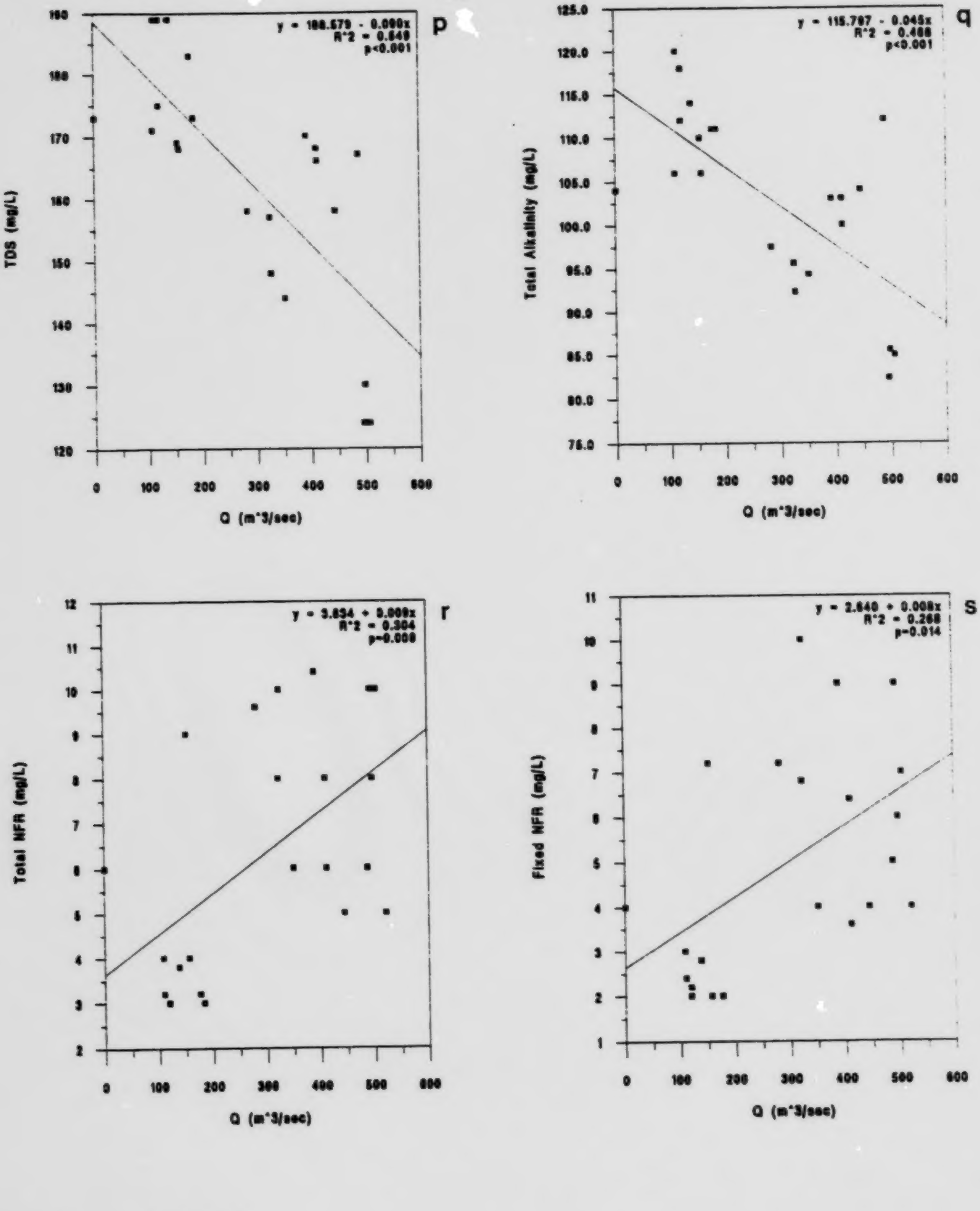

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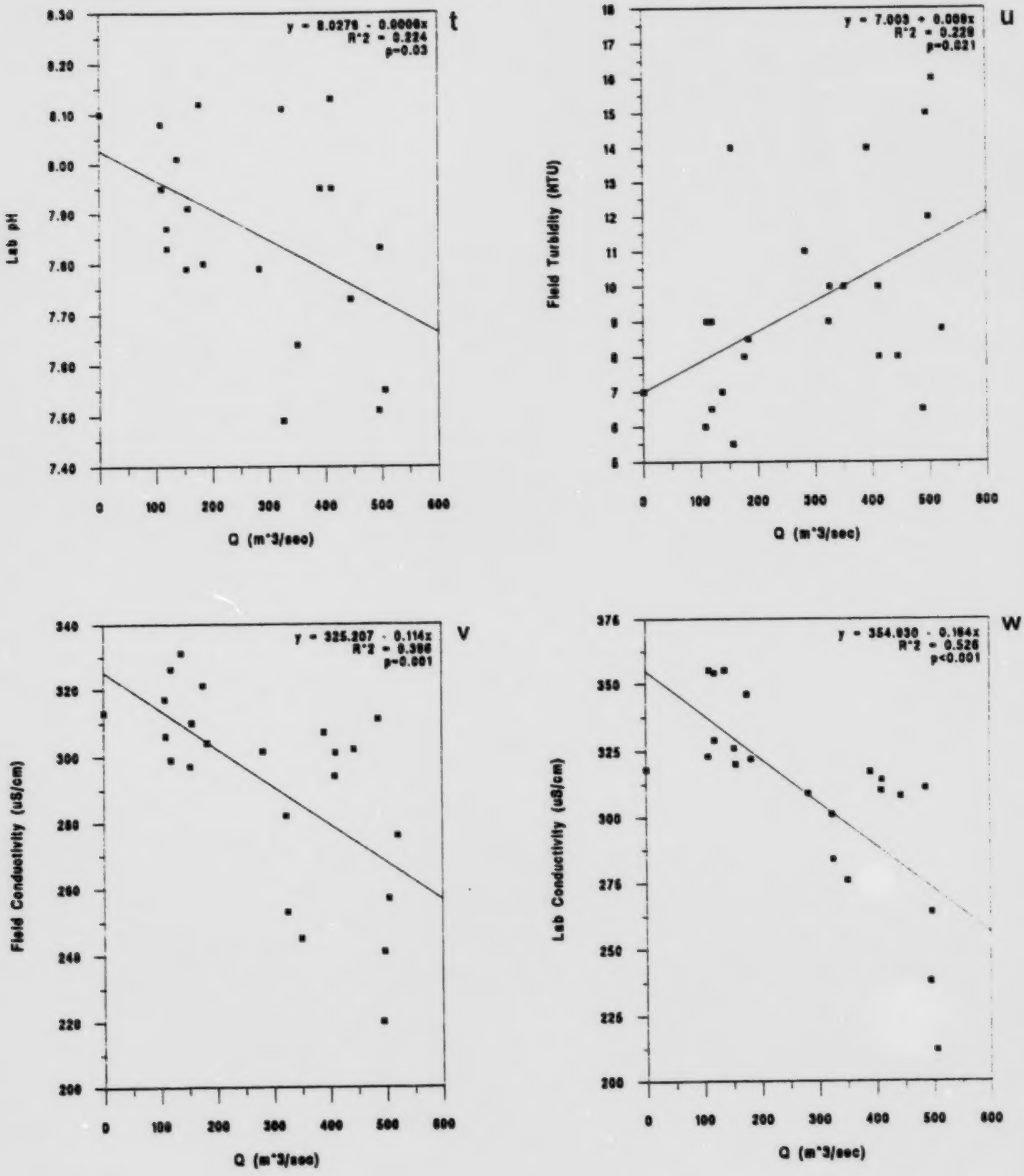


Figure 13. Continued.

LWR.

Most of the significant water quality - water discharge relationships at this site should be of some use for impact prediction and management purposes. All of the 23 significant correlations accounted for at least 21% of the variance in the dependent variable and all but two of the correlations (dissolved NO_2-NO_3 and total NH_3) were a reflection of a general linear trend in the data. The most useful discharge correlations are those involving conductivity (lab), TDS, alkalinity (total and bicarbonate), hardness (total and non-carbonate), dissolved Cl and SO_4, and dissolved Ca, Mg, K, and Na in which more than 40% of the variance was accounted for by the regression.

Duncan and Williamson (1988) examined the linear relationships between discharge and water chemistry in the Nelson River at Norway House, about 20 km upstream of Sea River Falls. As in the present study, they found conductivity, alkalinity, Ca, and Mg were negatively correlated and turbidity was positively correlated with discharge. Contrary to the present study, pH was found to be positively correlated with discharge. There is no clear reason for this discrepancy between the studies.

3.2.1.3 Seasonal Differences

The range of values in the ice-cover season overlapped the range in the ice-free season for all but 4 variables (total and fixed NFR, dissolved Mg, and reactive Si) during the 1987-1989 FEMP study period. Total and fixed NFR and reactive Si were lower under ice cover while Mg was higher (Figure 14). Three of the variables were correlated with discharge, NFR (total and fixed) positively and Mg negatively (Figure 13). With discharge at Sea River Falls typically lower under ice cover (220 to 232 m^3 sec^{-1}, Dec. to Mar.) than in the open water season (297 to 527 m^3 sec^{-1}, May to Nov., Water Survey of Canada 1990), these seasonal differences can be attributed to the fluctuations in discharge. Although the relationship with discharge generally accounts for the observed seasonal differences, it fails to account for the very few variables in which seasonal differences occurred given the large number of variables that were correlated with discharge. Discharge fluctuations also do not explain the summer maximum in reactive Si at this site. Release or resuspension from upstream lake sediments (Wetzel 1983), or contributions in eroded bank and shoreline materials are possible explanations, supported by the summer peak in NFR, although no specific site can be identified with the available information.

The pre-development dataset was comprised of 5 to 6 measurements between June 1971 and May 1972, with 2 to 3 of these under ice-cover. This dataset was judged to be inadequate for assessment of the occurrence of seasonal patterns.

3.2.1.4 Changes in Water Quality Since Lake Winnipeg Regulation

The pre-development data record for this site did not meet the minimum requirements for a statistical comparison, as noted above. Qualitative comparison indicates no marked change in physical and

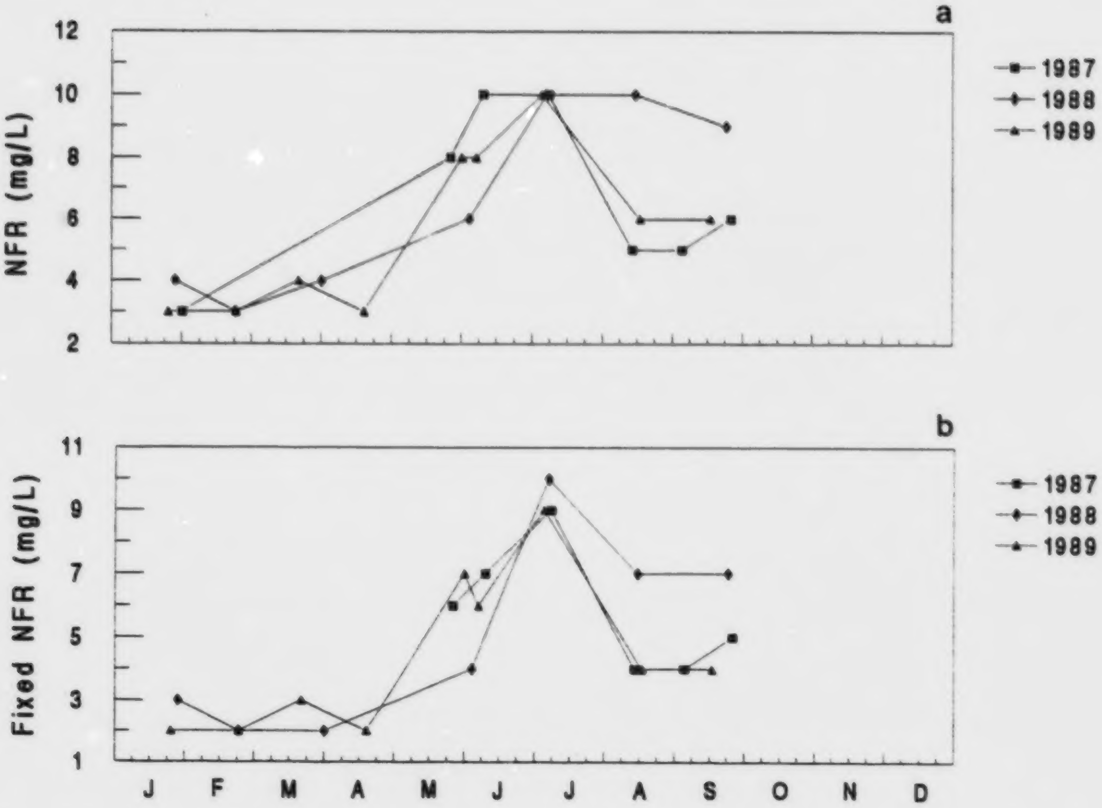


Figure 14. Seasonal water quality trends in the Nelson River below Sea River Falls. a. total NFR b. fixed NFR c. dissolved Mg d. reactive Si.

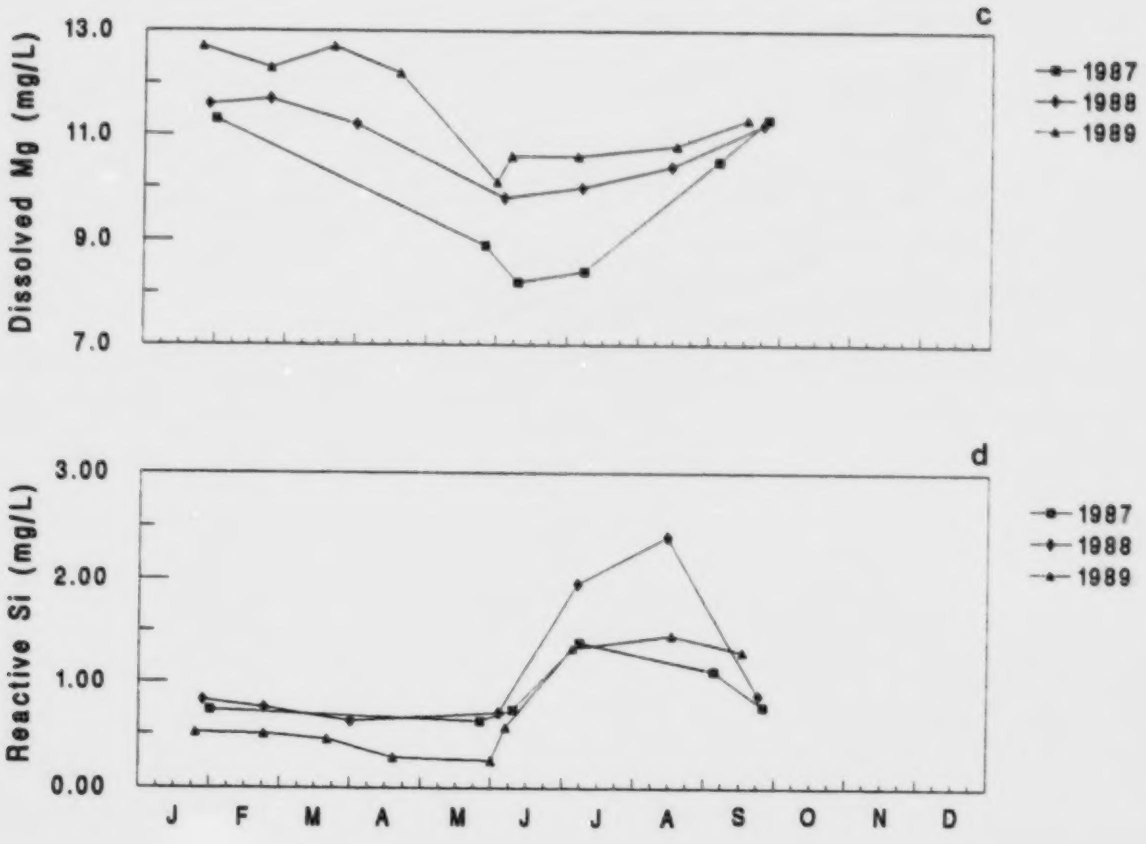


Figure 14. Continued.

chemical water quality. For the majority of variables (turbidity, TOC, NO_2-NO_3, pH, NFR (total and fixed), hardness, Na, K, and dissolved Pb), the pre-development mean is within 1 standard deviation (s.d.) of the post-development mean (Tables 12 and 13). The difference is less than 2 s.d. of the post-development mean for most other variables (conductivity, alkalinity, and reactive Si). Thus the pre-development data generally fall within the approximate 95% confidence limits for the post-development means, indicating no significant change. The only exceptions to this trend are dissolved Cu and Zn. Concentrations of these species prior to development are 5 to 10 times higher than the post-development means and are outside the range measured in the 1987-1989 sampling period. However, the pre-development data represent single analyses so it would be unwise to place much emphasis on these apparent changes.

Playle and Williamson (1986) found a number of small changes in their more extensive database for the Nelson River near Norway House. Total organic and inorganic carbon, total P, TKN, SO_4, and turbidity were 16-36% higher between mid-1876 and 1984 than from 1971 to mid-1976. These changes were generally attributed to shoreline degradation caused by level regulation and channel construction.

The only way of assessing the impacts of LWR regulation on water quality at Sea River Falls, in the absence of sufficient pre-development data for the site, is to use pre-development data for some other adequately sampled site. The Nelson River at Norway House is the only possible choice, as it was routinely sampled from

Table 13. Summary statistics for physical and chemical water quality variables measured in the Nelson River below Sea River Falls; June, 1971, to May, 1972. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Lab		7.7	0.2	7.9	7.4
Conductivity	Lab	$\mu S\ cm^{-1}$	252	46.2	307	182
Turbidity	Lab	NTU	9.2^+	3.6	13.0^+	2.8
NFR	Total	mg L^{-1}	7			
NFR	Fixed	mg L^{-1}	2			
Alkalinity	Total	mg L^{-1}	85.5	13.5	103	66.1
Hardness	Total	mg L^{-1}	103.5	17.0	129.0	81.4
Nitrogen	NO_2-NO_3	mg L^{-1}	0.09	0.08	0.14	0.03
Carbon	TOC	mg L^{-1}	8.5	3.5	12	5
K	Dissolved	mg L^{-1}	2.2	0.5	2.9	1.7
Na	Dissolved	mg L^{-1}	13.3	2.2	16.0	10.5
Si	Reactive	mg L^{-1}	1.9	0.4	2.4	1.5
Cu	Dissolved	mg L^{-1}	0.01			
Pb	Dissolved	mg L^{-1}	<0.001			
Zn	Dissolved	mg L^{-1}	0.011			
Discharge	Daily	$m^3\ sec^{-1}$	476	174	688	323

* Exceeds limit for drinking water.

1972 to 1974.

In order to use data from a surrogate site, it is necessary to demonstrate that water quality at the two sites should have been the same prior to development. This probably was the case for the Norway House and Sea River Falls sites provided there were no natural sites of erosion, significant local tributaries, or man-made structures or facilities between the sites which might contribute to significant differences. This validation would require some field reconnaissance and consultation with individuals knowledgeable in the recent history of development (i.e., past 25 years) along this reach of the Nelson River. Such a validation process was outside the scope of the present study, but should be considered as part of any subsequent investigation.

3.2.2 Jack River

3.2.2.1 Current Water Quality

Physical and chemical water quality in the Jack River was generally good, with levels for most parameters well below the Canadian guidelines for human consumption, recreational use, and the protection of aquatic life (Table 14). As at most other sites in the FEMP study, there were a number of exceptions to this trend. The mean extractable Al and total Cu concentrations exceeded the guideline levels for protection of aquatic life, as did the peak concentrations of total Pb. Mean turbidity exceeded the drinking water guideline as did the maximum extractable iron concentrations.

Table 14. Summary statistics for physical and chemical water quality variables measured in the Jack River (West Channel); January, 1987, to September, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.8	0.3	8.4	7.4
	Lab		7.90	0.19	8.19	7.44
Conductivity	Field	$\mu S\ cm^{-1}$	293	25	330	237
	Lab		314	27	349	245
Turbidity	Field	NTU	10^{+}	6	34^{+}	5
	Lab		9^{+}	5	26^{+}	3
Colour	True	Relative	8	4	15	3
TDS		$mg\ L^{-1}$	165	16	185	122
NFR	Total	$mg\ L^{-1}$	7	4	20	2
	Fixed	$mg\ L^{-1}$	5	4	17	1
Alkalinity	Total	$mg\ L^{-1}$	103.7	9.9	117.0	77.4
	Bicarbonate	$mg\ L^{-1}$	124.3	15.7	142.6	76.4
Hardness	Total	$mg\ L^{-1}$	118.5	10.1	133.5	93.1
	Non-carbonate	$mg\ L^{-1}$	14.7	3.5	20.4	9.1
Free CO_2		$mg\ L^{-1}$	3.38	2.27	8.01	0
Phosphorus	Particulate	$mg\ L^{-1}$	0.012	0.007	0.032	<0.006
	Ortho PO_4	$mg\ L^{-1}$	<0.003	0.002	0.009	<0.003
	Total Dissolved	$mg\ L^{-1}$	0.010	0.005	0.025	0.004
	Total	$mg\ L^{-1}$	0.022	0.006	0.037	0.015
Nitrogen	Particulate	$mg\ L^{-1}$	0.050	0.026	0.101	<0.010
	Total Dissolved	$mg\ L^{-1}$	0.300	0.045	0.470	0.230
	NO_2-NO_3	$mg\ L^{-1}$	0.011	0.011	0.048	<0.010
	Total NH_3	$mg\ L^{-1}$	0.025	0.017	0.070	0.006
	Total	$mg\ L^{-1}$	0.35	0.05	0.50	0.28
Carbon	DOC	$mg\ L^{-1}$	7.30	0.46	8.10	6.59
	POC	$mg\ L^{-1}$	0.387	0.162	0.753	0.150
	TOC	$mg\ L^{-1}$	7.68	0.43	8.59	7.12
	DIC	$mg\ L^{-1}$	24.8	2.5	28.1	18.3
Chlorophyll-a		$mg\ L^{-1}$	0.002	0.002	0.005	<0.001
Ca	Dissolved	$mg\ L^{-1}$	29.5	2.4	33.0	23.6
Mg	Dissolved	$mg\ L^{-1}$	10.9	1.1	12.6	8.3
K	Dissolved	$mg\ L^{-1}$	2.30	0.21	2.71	1.87
Na	Dissolved	$mg\ L^{-1}$	16.2	2.2	19.4	11.3

Table 14. continued

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	0.70	0.46	1.91	0.23
Cl	Dissolved	mg L^{-1}	18.6	2.9	23.3	12.8
F	Dissolved	mg L^{-1}	0.11	0.02	0.16	0.08
SO_4	Dissolved	mg L^{-1}	24.4	3.3	29.0	17.4
Al	Extractable	mg L^{-1}	0.176^{+++}	0.126	0.442^{+++}	<0.100
	Dissolved	mg L^{-1}	<0.100		<0.100	<0.100
As	Dissolved	mg L^{-1}	0.0006	0.0001	0.0008	0.0004
Ba	Total	mg L^{-1}	<0.06	0.01	0.06	<0.06
Cd	Total	mg L^{-1}	0.0001	0.0001	0.0004	<0.0001
	Dissolved	mg L^{-1}	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0001	0.0007	<0.0005
Cu	Total	mg L^{-1}	0.0034^{+++}	0.0038	0.0159^{+++}	0.0011
	Dissolved	mg L^{-1}	0.0021	0.0013	0.0061	0.0013
Fe	Extractable	mg L^{-1}	0.187	0.106	0.457$^{+,+++}$	0.070
	Dissolved	mg L^{-1}	0.025	0.012	0.052	0.004
Pb	Total	mg L^{-1}	0.0007	0.0006	0.0027^{+++}	<0.0007
	Dissolved	mg L^{-1}	0.0017	0.0025	0.0077	<0.0007
Mn	Extractable	mg L^{-1}	0.012	0.007	0.034	<0.002
	Dissolved	mg L^{-1}	0.003	0.002	0.006	<0.002
Ni	Total	mg L^{-1}	0.0012	0.0004	0.0020	0.0005
	Dissolved	mg L^{-1}	0.0010	0.0006	0.0030	<0.0005
Se	Dissolved	mg L^{-1}	0.0002	0.0001	0.0004	<0.0001
V	Total	mg L^{-1}	0.0007	0.0004	0.0017	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0003	0.0014	<0.0005
Zn	Total	mg L^{-1}	0.0032	0.0031	0.0131	0.0005
	Dissolved	mg L^{-1}	0.0013	0.0015	0.0045	<0.0003

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recreational use.

$^{+++}$ Exceeds limit for protection of aquatic life.

3.2.2.2 Seasonal Differences

There were few clear differences in water quality between the ice-free and ice-cover seasons during the 1987-1989 FEMP study. NFR was lower under ice-cover, as was chlorophyll a, while dissolved Cd and Ni concentrations were higher under ice cover (Figure 15). The lower chlorophyll a under ice is consistent with the trend at other sites. The lower NFR and higher Cd and Ni under ice could be a product of lower winter flows, but this cannot be confirmed due to the lack of discharge data for the site.

3.2.2.3 Water Quality Changes Since Lake Winnipeg Regulation

There are no pre-development water quality data for this site and the immediate post-development data record amounts to a pair of samples collected in November, 1976, and June, 1977. Consequently there is no means of directly determining to what extent, if any, water quality has changed since Lake Winnipeg regulation. It may be possible to use pre-development data for another upstream site as a surrogate, as suggested for the site at Sea River Falls. The only upstream site which was adequately sampled prior to development is the Nelson River at Warren Landing, which has been sampled since 1969. Verification of this site as a suitable surrogate, as described above for the Sea River Falls surrogate, would also be required. Again, this verification process was outside the scope of the present study.

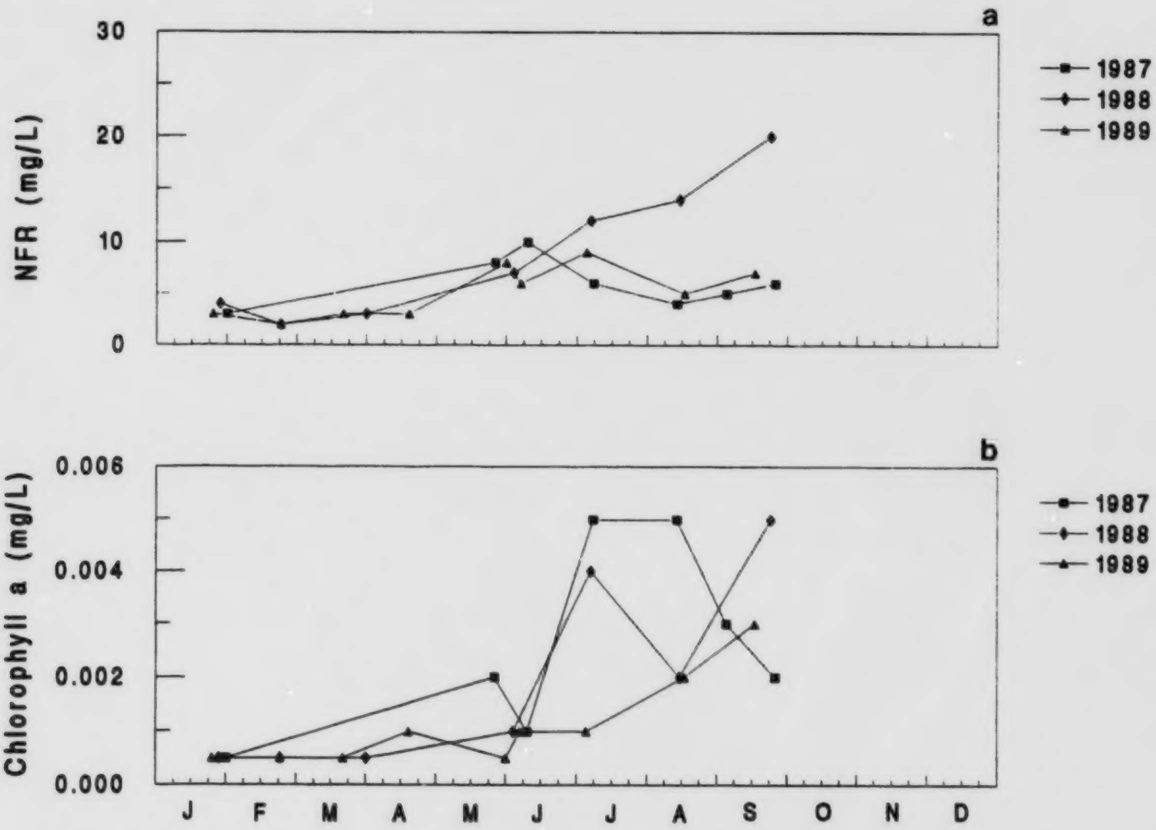
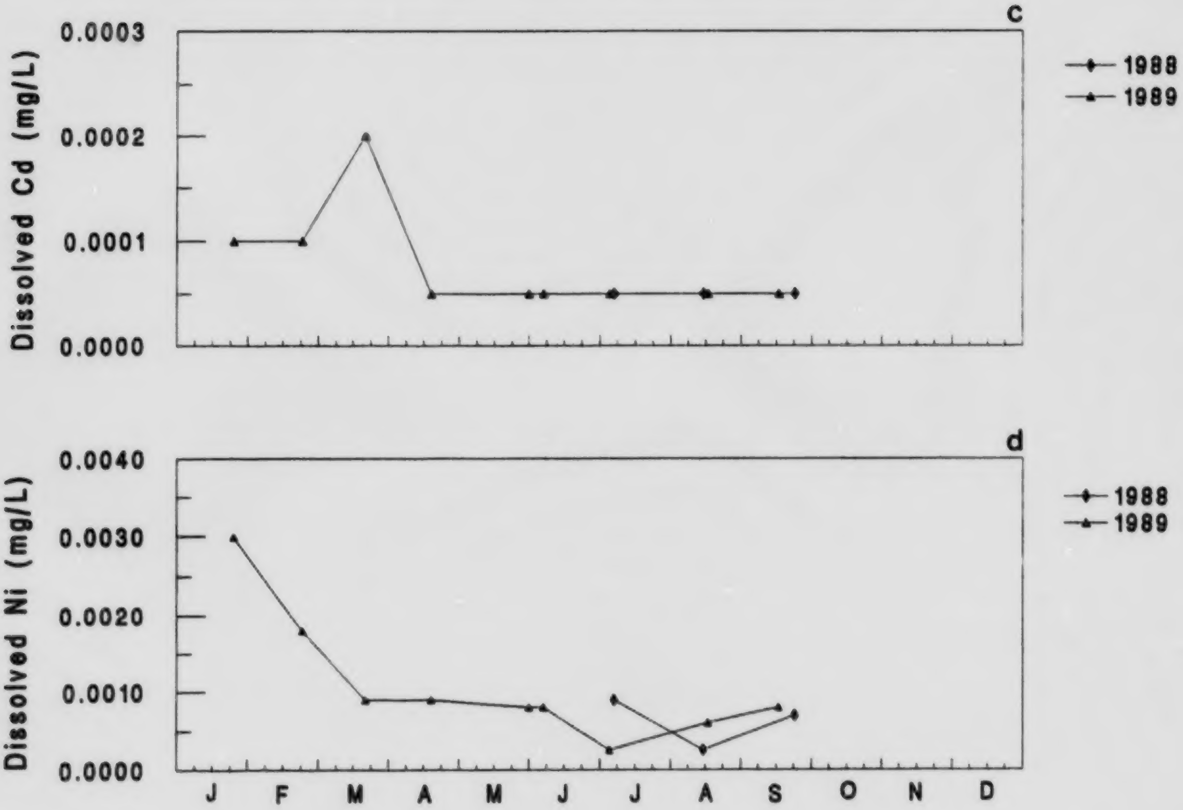


Figure 15. Seasonal water quality trends in the Jack River (West Channel) near Norway House. a. total NFR b. chlorophyll a c. dissolved Cd d. dissolved Ni.



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3.2.3 Bacterial Studies

3.2.3.1 Norway House

Geometric mean fecal coliform bacterial counts for the seasonal study ranged from 3 per 100 mL at Historic Dock by Bay Store Bridge to 39 per 100 mL at Government Dock (Figure 16). Counts in samples from the Sewage Outlet Bay at Church Point, Green Dock, School Dock, Asiniwapiska, Bridge into Fort Island, and Historic Dock by Bay Store Bridge were consistently below 200 per 100 mL. Counts in 3 of the 4 samples from Government Dock also were well below 200 per 100 mL, but the September 22 count was 2230.

Intensive studies of bacterial water quality were conducted in the vicinity of the sewage and water treatment plants on August 11 and 24, 1987. On the first survey, fecal coliform counts ranged between 0 and 11 bacteria per 100 mL with a geometric mean in the 10 samples of 5 per 100 mL (Figure 17a). Counts ranged between 0 and 567 bacteria per 100 mL on the second survey with a geometric mean of 15 per 100 mL (Figure 17b). The overall geometric mean count for both intensive surveys was 9 fecal coliform bacteria per 100 mL. These analyses indicate bacteriological water quality in the vicinity of the water and sewage treatment plants is excellent for recreational uses, with a geometric mean fecal coliform count just 5% of the Canadian guideline of 200 bacteria per 100 mL. The regular samples also indicate bacterial water quality is generally within the Canadian guideline. It is not known if water quality at all sites is in strict compliance with the guideline, as the sampling frequency did not meet the 5 samples per 30 days specified

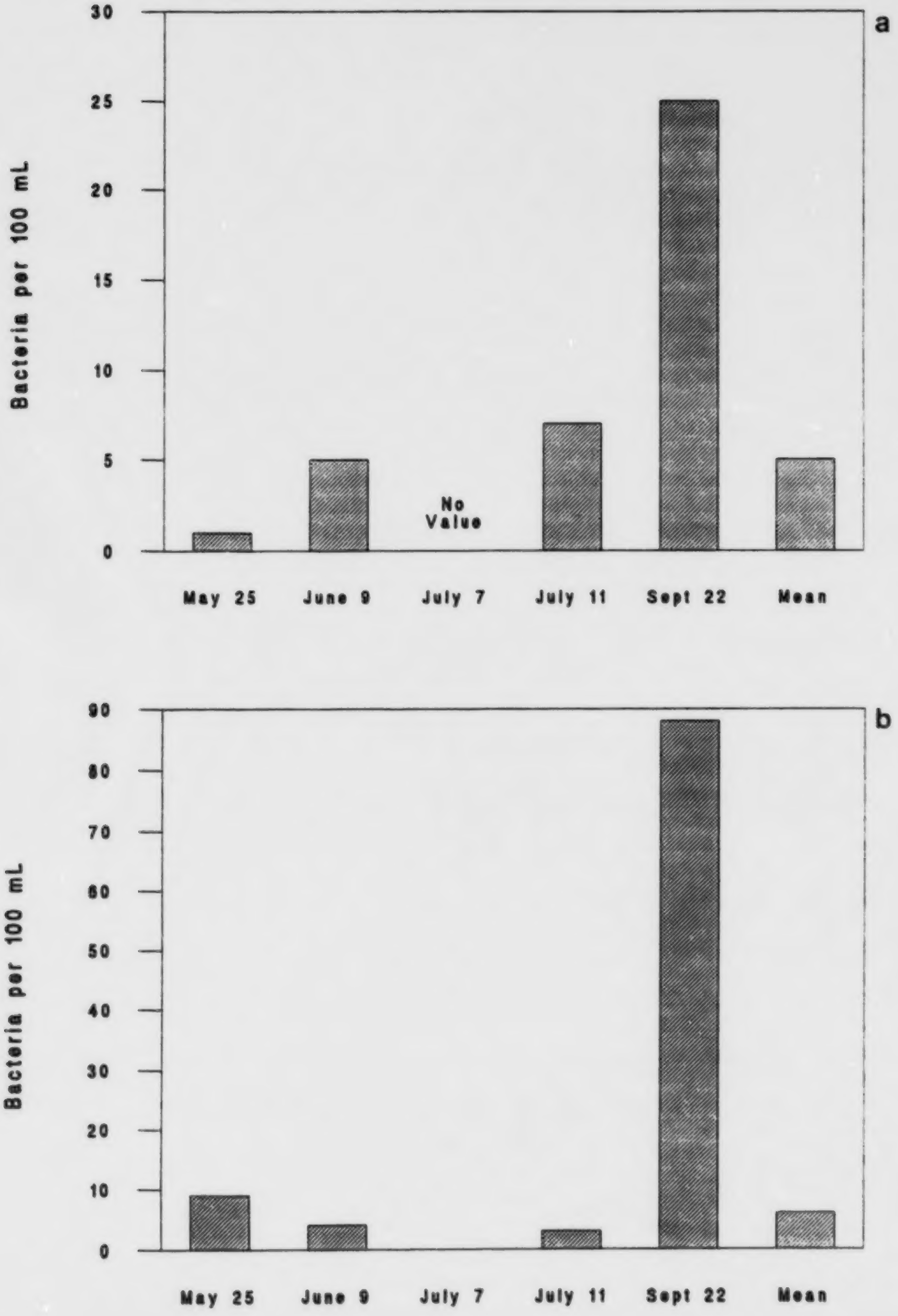
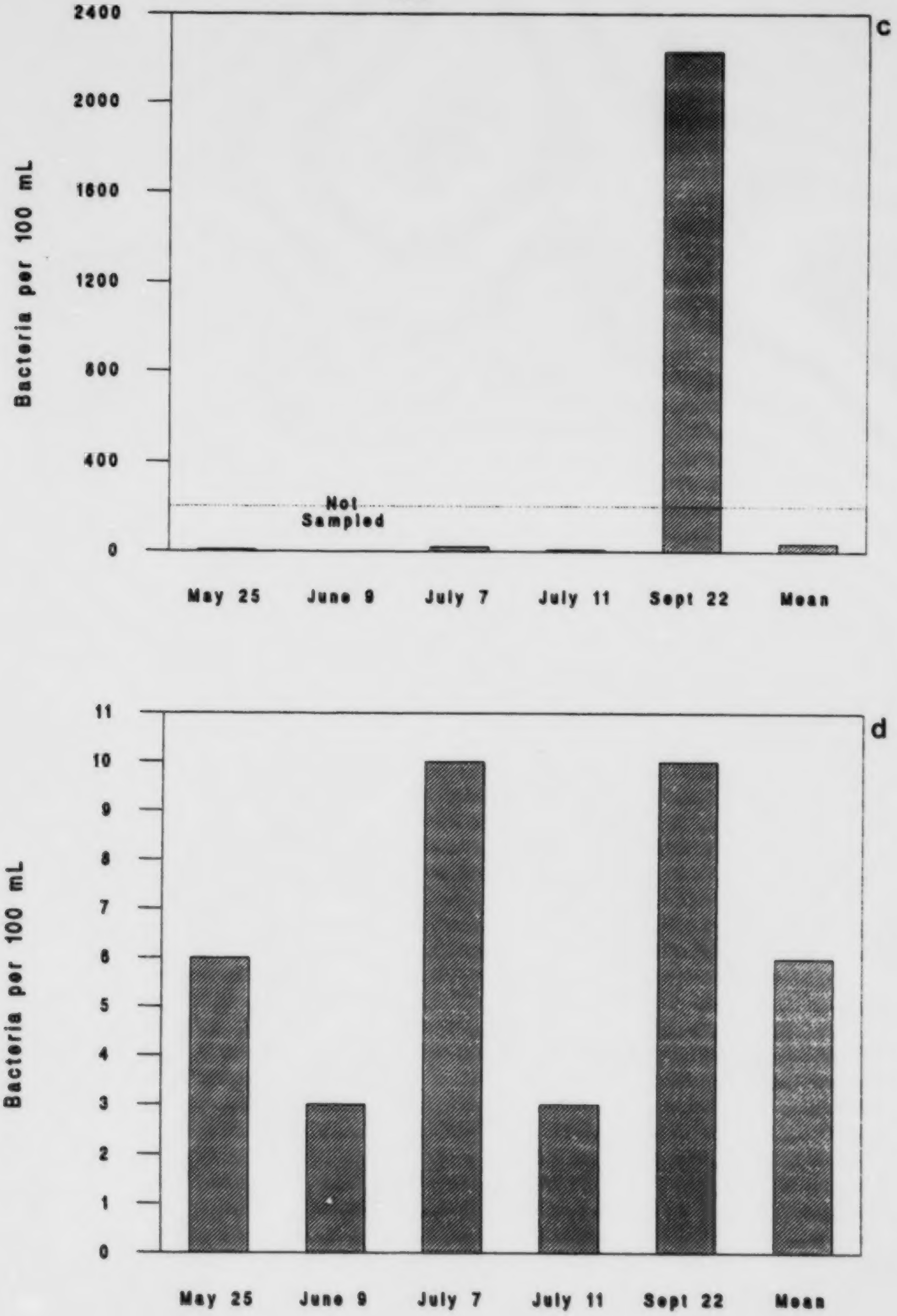


Figure 16. Fecal coliform bacteria counts from the seasonal study in the vicinity of Norway House. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Asiniwapiska b. Bridge into Fort Island c. Government Dock d. Green Dock e. Historic Dock at Bay Store Bridge f. School Dock g. Sewage Outlet Bay at Church Point. See Appendix 1 for site locations and descriptions.



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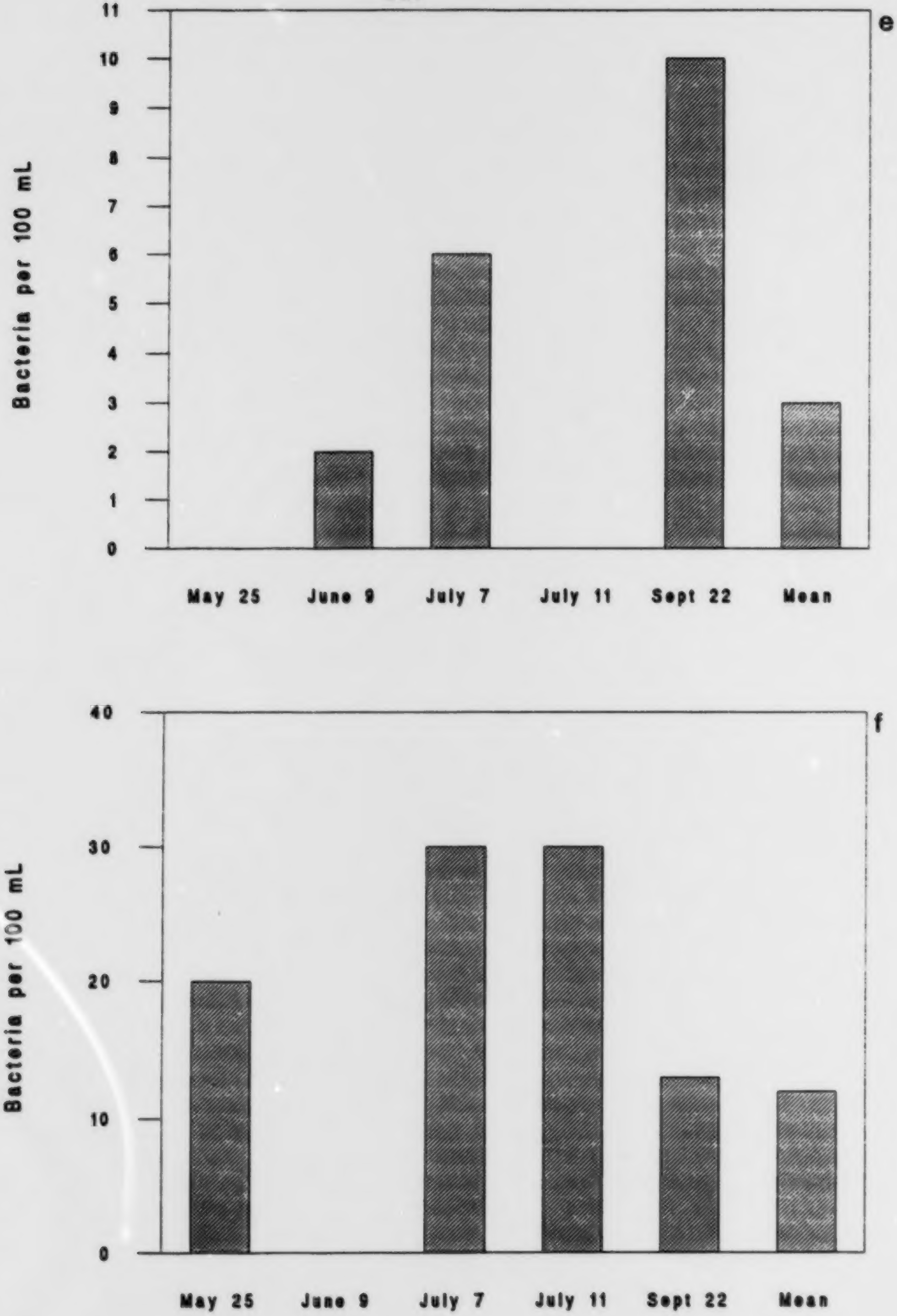

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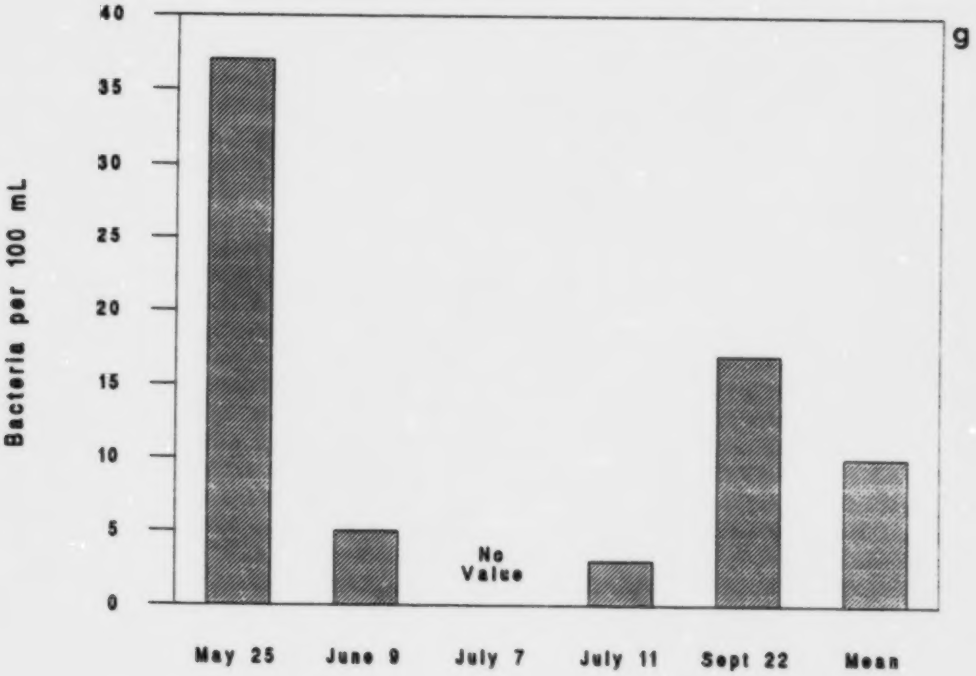

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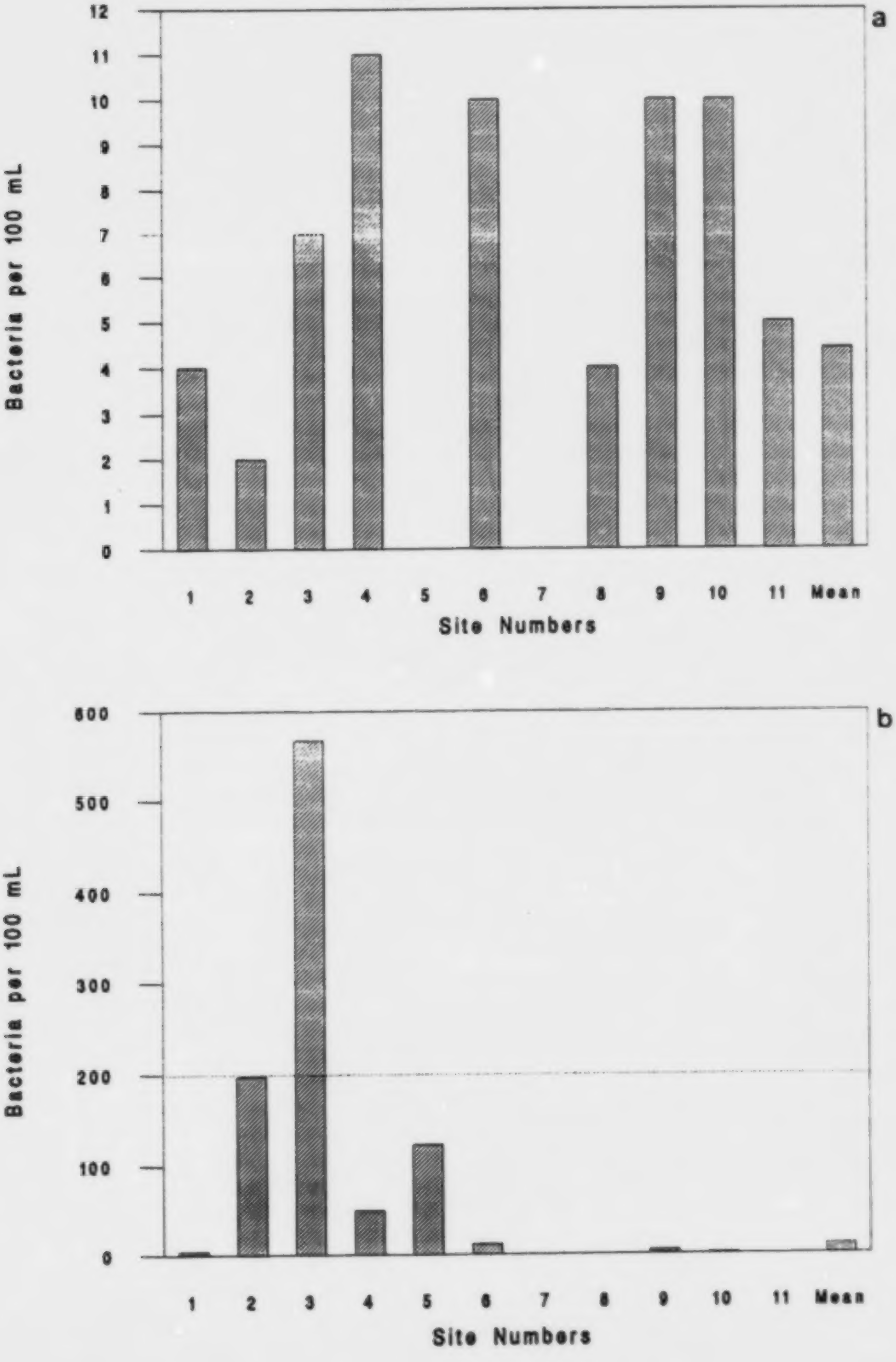


Figure 17. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Norway House Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 11 August 1987 b. Sampled 24 August 1987. See Appendix 1 for site descriptions and locations.

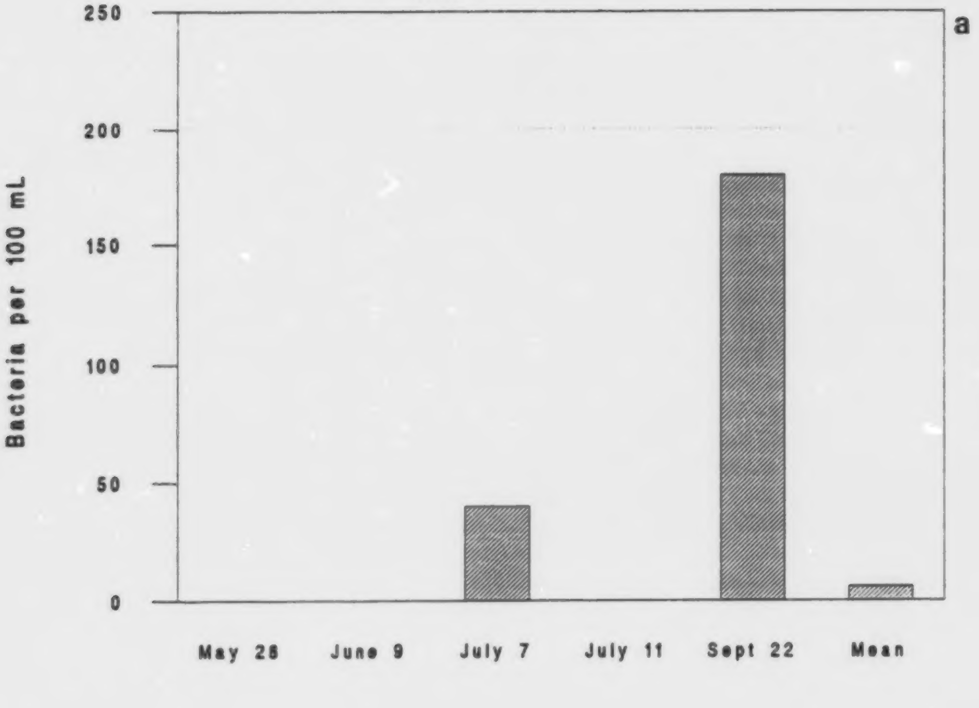
under the guideline. Future studies should ensure the designated sampling frequency is met at all sites.

3.2.3.2 Cross Lake

Geometric mean fecal coliform bacteria counts for the seasonal study ranged from 6 per 100 mL at Beardy's Point to 44 at Wapak Swimming Area (Figure 18). Fecal coliform counts in water samples from Saggitawak-Swimming Area and Beardy's Point were consistently below 200 per 100 mL. One of the five samples from each of the Saggitawak Bridge, Wapak Swimming Area, and Water Treatment Plant Intake exceeded 200 per 100 mL.

Intensive studies of bacterial water quality in the vicinity of the water treatment plant intake were conducted on August 11 and 25, 1987. Fecal coliform counts were below 10 per 100 mL in 14 of the 18 samples, and counts exceeded 200 per 100 mL in just 2 samples (Figure 19). The geometric mean of the counts on the August 11 survey was 21 per 100 mL compared with 4 per 100 mL on the August 25 survey. The overall geometric mean for both surveys was 9 counts per 100 mL. The two high counts, 1030 and 2280, occurred at sites 6 and 8, respectively on August 11. By August 25, counts at these sites had dropped to 1 and 7 bacteria per 100 mL, indicating the high bacterial numbers were relatively transient.

On the basis of the intensive studies, bacteriological water quality in the area of the water treatment plant intake can be classified as excellent for recreational purposes, with a geometric mean fecal coliform count for the study less than 10% of the



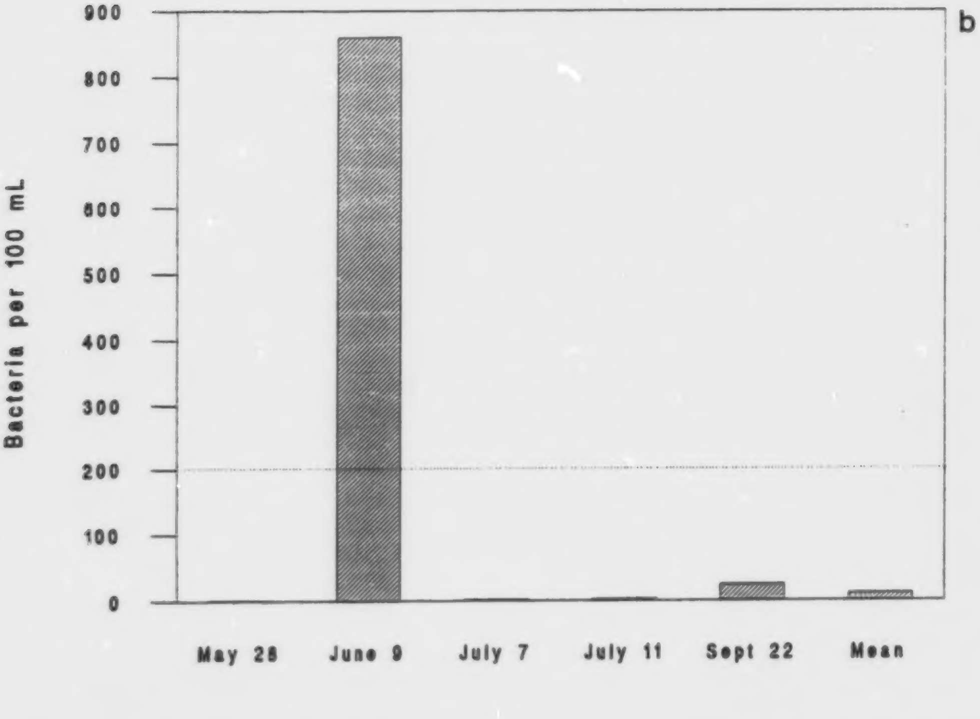
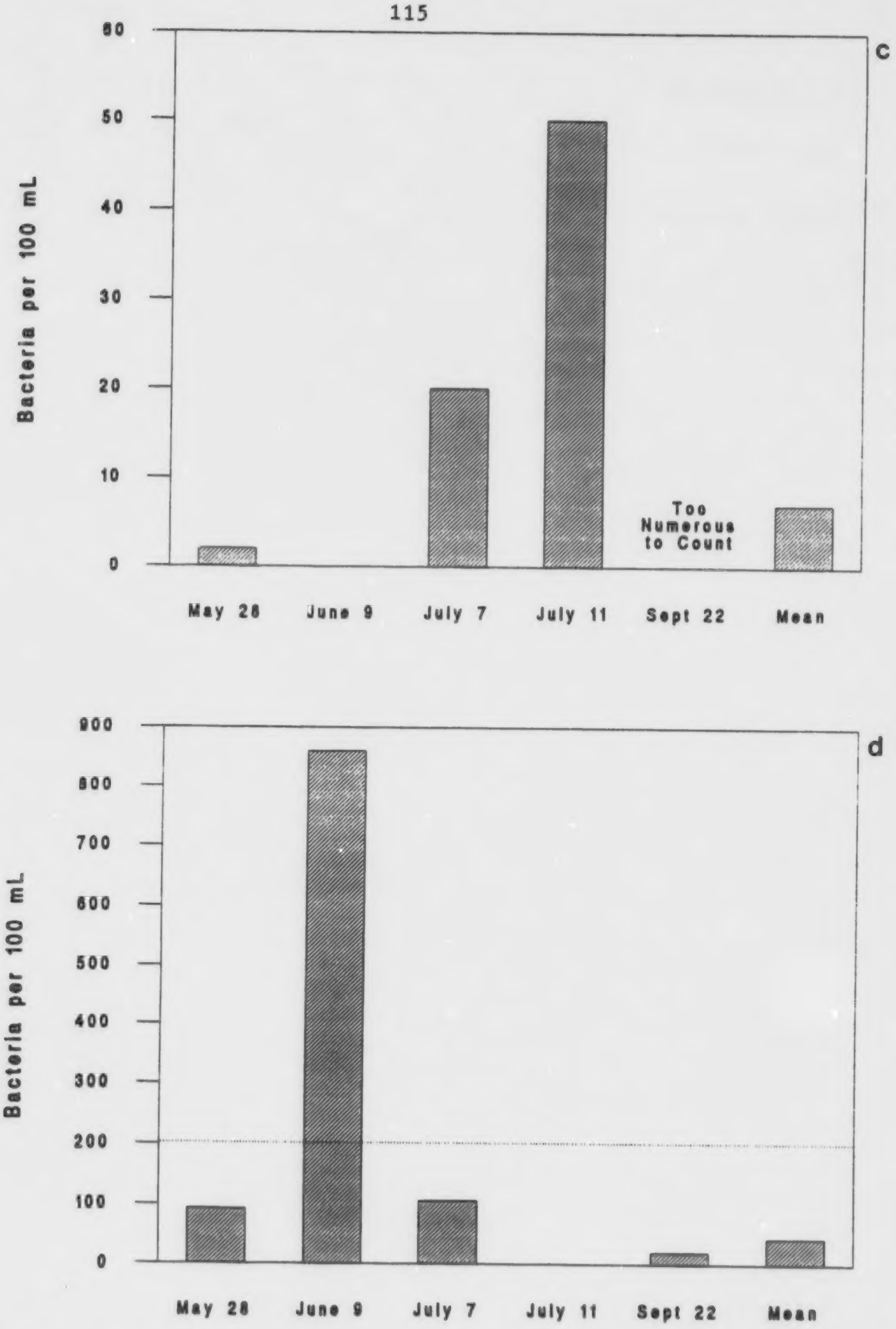


Figure 18. Fecal coliform bacteria counts from the seasonal study in the vicinity of Cross Lake. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Beardy's Point b. Saggitawak Bridge c. Saggitawak Swimming Area d. Wapak Swimming Area e. Left of the Water Treatment Plant Intake. See Appendix 1 for site descriptions and locations.



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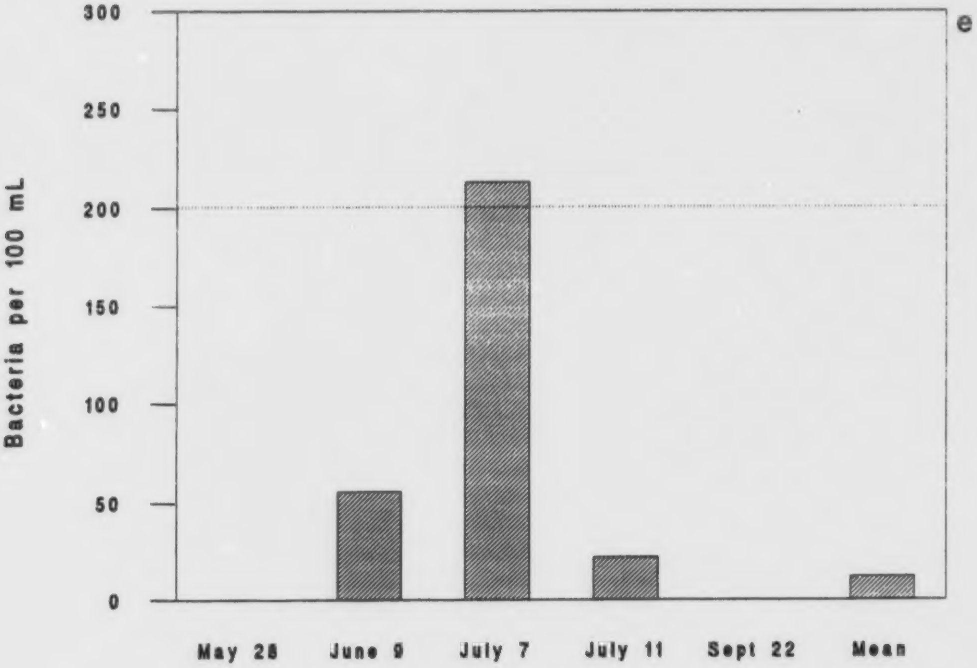

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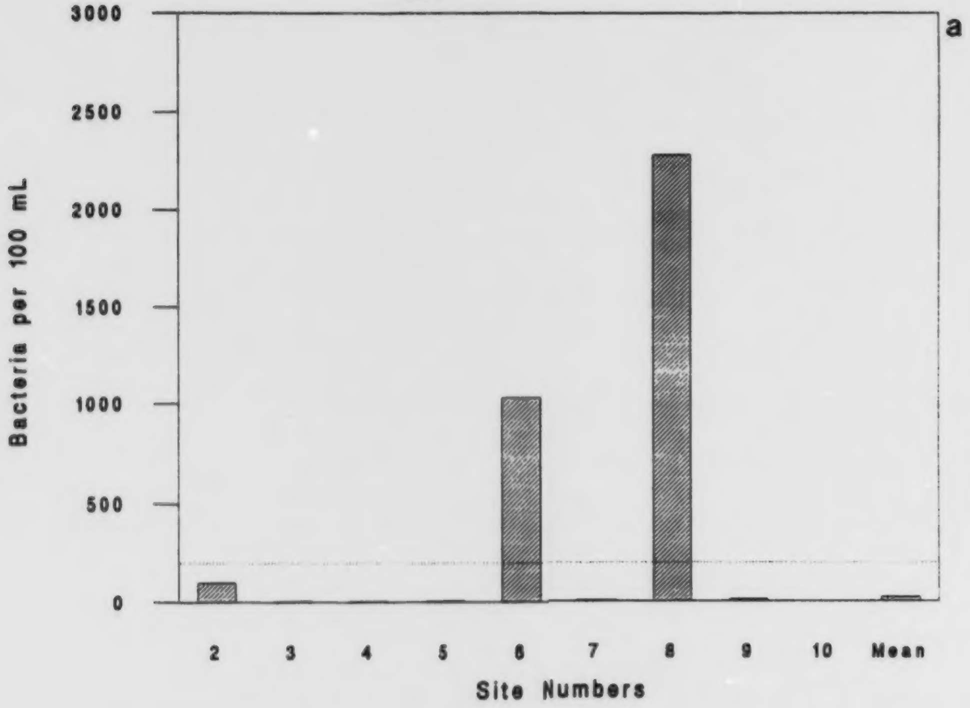


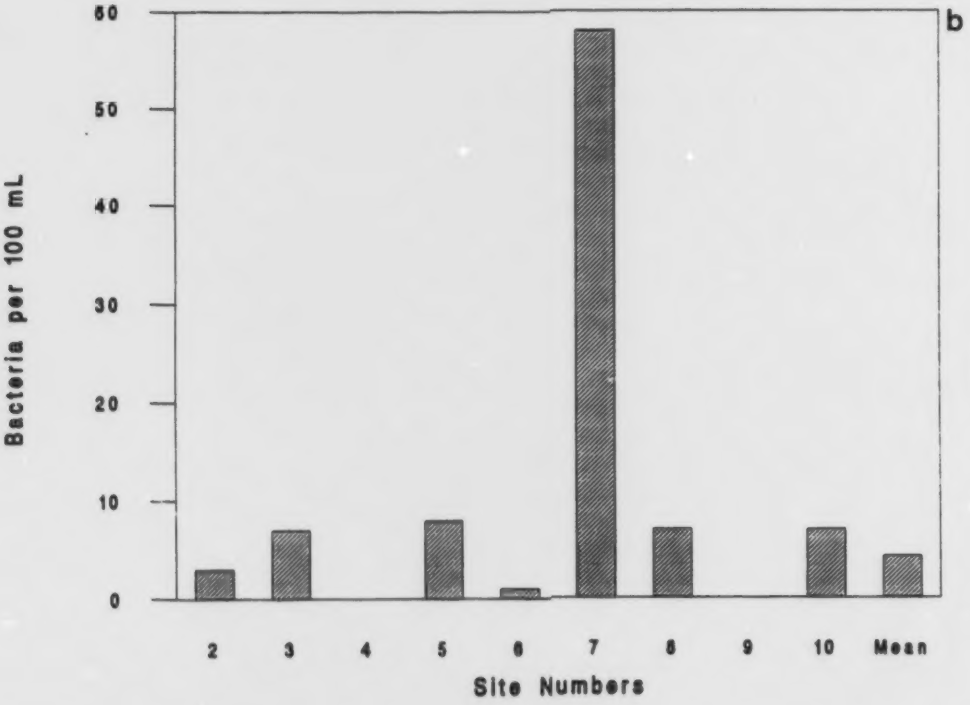


Figure 19. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Cross Lake Water Treatment Plant Intake. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 11 August 1987 b. Sampled 25 August 1987. See Appendix 1 for site descriptions and locations.

Canadian guideline of 200 bacteria per 100 mL. The results of the seasonal study indicate recreational waters in the Cross Lake community appear to be of good quality for recreational uses. The occasional high bacterial count should not be considered as evidence of contamination as the confidence interval for an estimate based on a single sample is very broad. Williamson (1988a) indicates a count of 1000 bacteria per 100 mL in a single sample is within the 90% confidence interval for a true bacterial density of 200 per 100 mL. Sampling was not of sufficient frequency in the seasonal study to determine if quality was in strict compliance with the Canadian guideline. Future studies should ensure that at least five samples are collected within a 30 day period at each site of interest.

3.3 Threepoint Lake and Nelson House Area

3.3.1 Rat River above Threepoint Lake

3.3.1.1 Current Water Quality

Water quality in the Rat River was generally similar to that in the natural rivers. Among the variables for which water quality guidelines have been established, most were consistently well below the guideline values (Table 15). The exceptions generally were the same as in the natural rivers. Turbidity always exceeded the drinking water guideline and peak levels (48 NTU) approached the recreational use guideline. Mean colour was at the high end of the acceptable range for untreated drinking water, with peak values exceeding the guideline. Extractable Al concentrations always

Table 15. Summary statistics for physical and chemical water quality variables measured in the Rat River above Threepoint Lake; September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.3	7.9	7.0
	Lab		7.67	0.17	8.15	7.29
Conductivity	Field	µS cm^{-1}	99	10	115	79
	Lab		102	7	120	86
Turbidity	Field	NTU	19^{+}	8	48^{+}	9^{+}
	Lab		17^{+}	8	46^{+}	11^{+}
Colour	True	Relative	14	4	20^{+}	5
TDS		mg L^{-1}	53	4	62	46
NFR	Total	mg L^{-1}	10	7	42	5
	Fixed	mg L^{-1}	9	6	38	4
Alkalinity	Total	mg L^{-1}	46.4	3.1	53.2	40.8
	Bicarbonate	mg L^{-1}	56.6	3.8	64.9	49.7
Hardness	Total	mg L^{-1}	45.9	3.1	54.0	39.9
	Non-carbonate	mg L^{-1}	0.4	0.7	2.6	0
Free CO_2		mg L^{-1}	3.36	2.42	9.19	0
Phosphorus	Particulate	mg L^{-1}	0.016	0.009	0.039	<0.006
	Ortho PO_4	mg L^{-1}	0.003	0.003	0.012	<0.003
	Total Dissolved	mg L^{-1}	0.010	0.006	0.024	0.003
	Total	mg L^{-1}	0.026	0.008	0.049	0.013
Nitrogen	Particulate	mg L^{-1}	0.049	0.021	0.120	<0.010
	Total Dissolved	mg L^{-1}	0.235	0.043	0.330	0.170
	NO_2-NO_3	mg L^{-1}	0.019	0.025	0.086	<0.010
	Total NH_3	mg L^{-1}	0.016	0.011	0.042	<0.005
	Total	mg L^{-1}	0.28	0.05	0.44	0.22
Carbon	DOC	mg L^{-1}	6.04	0.73	8.80	3.79
	POC	mg L^{-1}	0.363	0.153	0.900	0.160
	TOC	mg L^{-1}	6.40	0.71	8.96	4.53
	DIC	mg L^{-1}	10.9	0.7	12.8	9.6
Chlorophyll-a		mg L^{-1}	0.002	0.001	0.004	<0.001
Ca	Dissolved	mg L^{-1}	12.3	1.0	14.2	10.4
Mg	Dissolved	mg L^{-1}	3.7	0.3	4.6	3.3
K	Dissolved	mg L^{-1}	1.02	0.07	1.24	0.93
Na	Dissolved	mg L^{-1}	2.7	0.3	3.5	2.1

Table 15. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	1.86	0.73	3.38	0.69
Cl	Dissolved	mg L^{-1}	1.1	0.2	1.6	0.7
F	Dissolved	mg L^{-1}	0.09	0.01	0.13	0.07
SO$_4$	Dissolved	mg L^{-1}	2.5	0.6	3.6	1.1
Al	Extractable	mg L^{-1}	0.357^{+++}	0.116	0.712^{+++}	0.161^{+++}
	Dissolved	mg L^{-1}	0.078	0.050	0.239	<0.100
As	Dissolved	mg L^{-1}	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L^{-1}	<0.08	0.02	0.13	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0004^{+++}	<0.0001
	Dissolved	mg L^{-1}	<0.0001	0.0001	0.0002	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0002	0.0012	<0.0005
Cu	Total	mg L^{-1}	0.0053^{+++}	0.0091	0.0400^{+++}	0.0005
	Dissolved	mg L^{-1}	0.0012	0.0010	0.0040	<0.0005
Fe	Extractable	mg L^{-1}	0.435$^{+,+++}$	0.131	0.926$^{+,+++}$	0.143
	Dissolved	mg L^{-1}	0.100	0.085	0.479	0.033
Pb	Total	mg L^{-1}	0.0008	0.0006	0.0021^{+++}	<0.0007
	Dissolved	mg L^{-1}	0.0008	0.0009	0.0035	<0.0007
Mn	Extractable	mg L^{-1}	0.019	0.013	0.081^{+}	0.004
	Dissolved	mg L^{-1}	0.003	0.004	0.016	<0.002
Ni	Total	mg L^{-1}	0.0011	0.0006	0.0032	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0003	0.0010	<0.0005
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0004	<0.0001
V	Total	mg L^{-1}	0.0009	0.0004	0.0018	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0002	0.0008	<0.0005
Zn	Total	mg L^{-1}	0.0049	0.0057	0.0184	0.0006
	Dissolved	mg L^{-1}	0.0016	0.0024	0.0081	<0.0003
Discharge	Daily	m^3 sec^{-1}	814	104	990	665

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recreational use.

$^{+++}$ Exceeds limit for protection of aquatic life.

exceeded the aquatic life guideline, by a factor of 3.5 on average and by as much as a factor of 7. Extractable Fe exceeded the drinking water and aquatic life guidelines in all samples; by about 45% on average. The total Cu concentration exceeded the aquatic life guideline by a factor of 2.7 on average, and by as much as a factor of 20 at peak concentrations. Peak concentrations of total Cd and total Pb also exceeded the respective aquatic life guidelines. Peak extractable Mn exceeded the drinking water objective.

3.3.1.2 Water Quality - Water Discharge Relationships

Positive correlations dominated the significant relationships between water quality and discharge at this station. These positive correlates included TDS, alkalinity (total and bicarbonate), total hardness, ortho PO_4, dissolved Ca and Se (Figure 20). Negative correlates were particulate N and POC.

The dominance of positive correlations at this site is the reverse of the situation in the natural rivers and in the Burntwood River at Thompson prior to diversion, where negative correlations dominated (Sections 3.1.3 and 3.4.2.1). This suggests negative correlations probably dominated at the Rat River site prior to diversion as well. The current nature of the discharge relationships most likely is a product of shoreline erosion along the lower Rat River.

None of these correlations is of much use for impact prediction or management. The regressions accounted for just 15 to

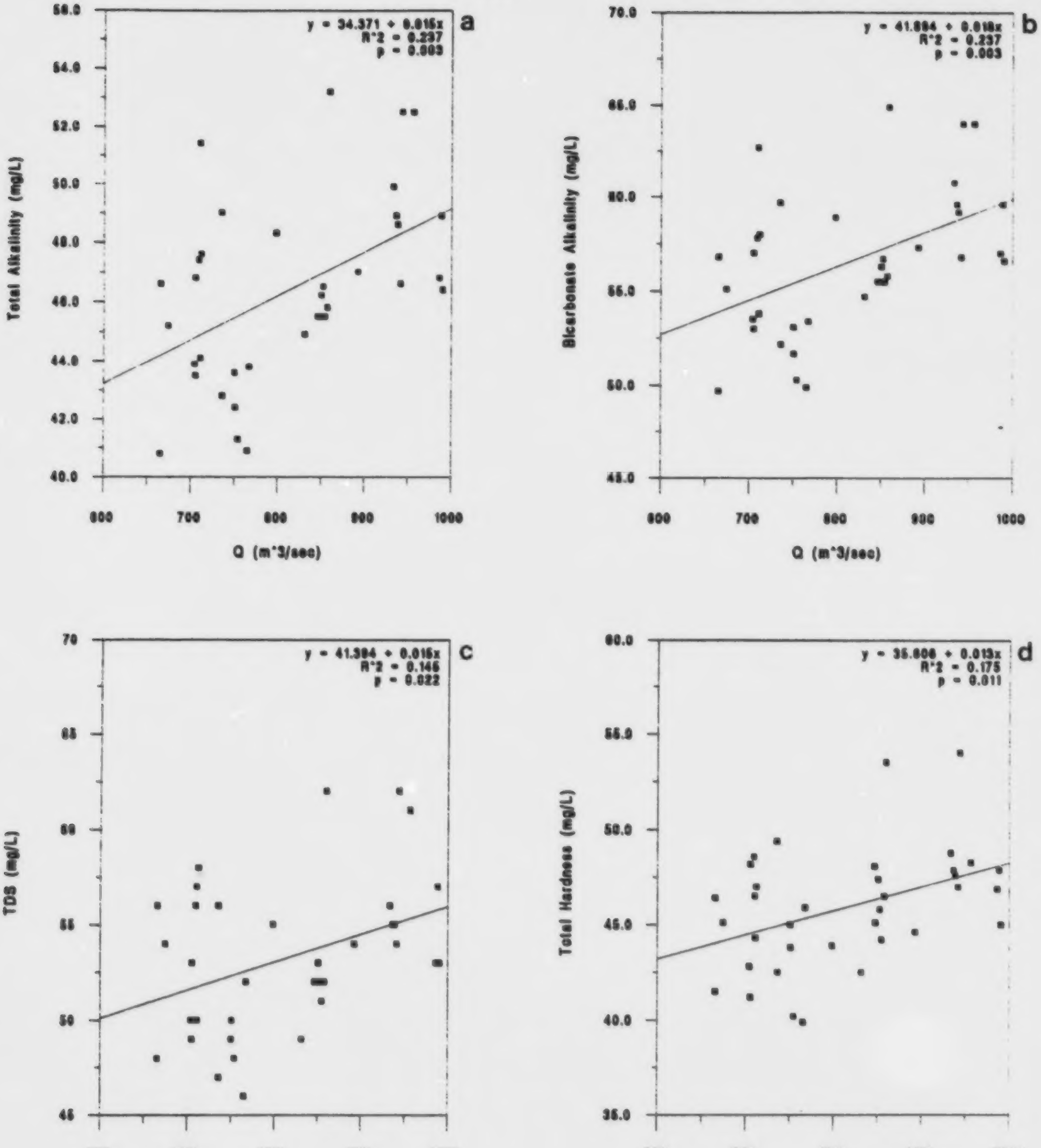
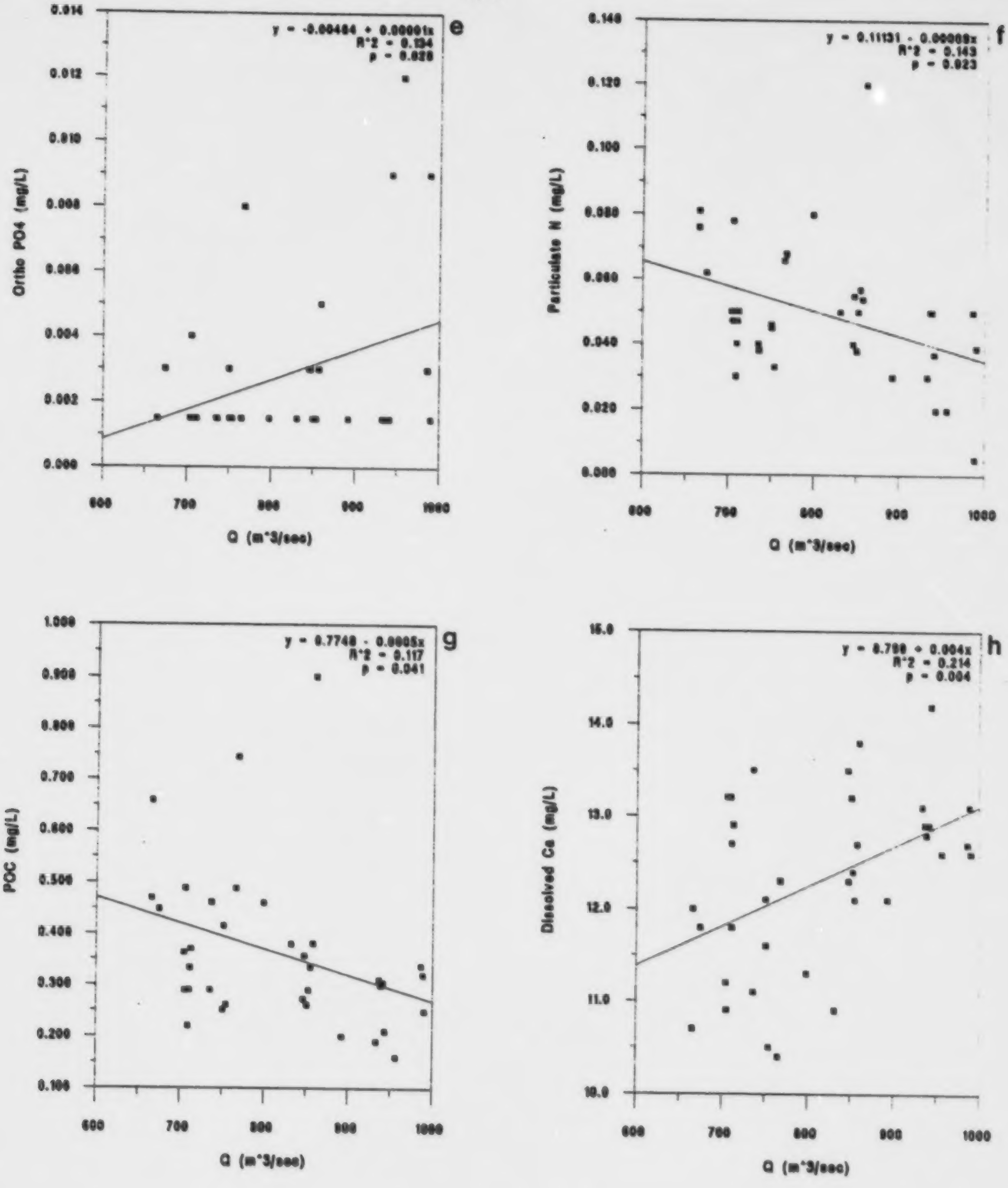


Figure 20. Water chemistry - water discharge relationships in the Rat River above Threepoint Lake, 1987-1989. a. total alkalinity b. bicarbonate alkalinity c. TDS d. total hardness e. ortho PO_4 f. particulate N g. POC h. dissolved Ca i. dissolved Se.



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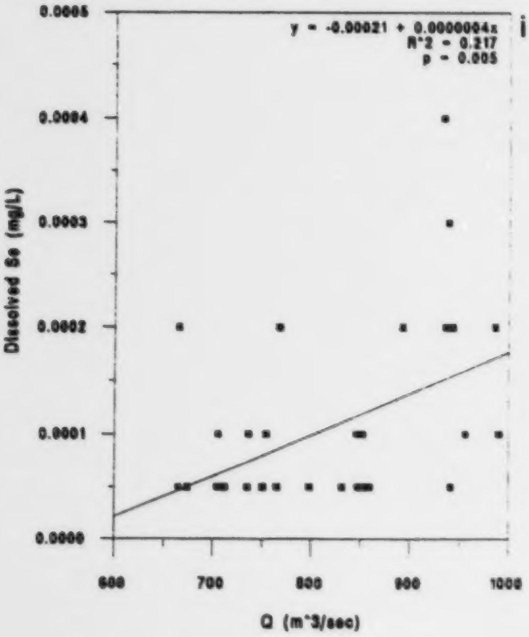

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25% of total variance in the dependent variables and none of the discharge-correlated variables was associated with adverse impacts either at this site or downstream.

3.3.1.3 Seasonal Differences

Few variables displayed distinct differences between the ice-cover and ice-free seasons during the FEMP study, although TDN, dissolved NO_2-NO_3, total N, and total Ni occurred in higher concentrations under ice while chlorophyll a concentrations were higher in the ice-free season (Figure 21). None of these variables was correlated with discharge, eliminating discharge fluctuations as a possible cause of the seasonality. The higher concentrations of total N and TDN in winter can largely be attributed to the higher dissolved NO_2-NO_3 levels. Reduced algal growth rates and standing crops under ice, and therefore lower dissolved N demand, can lead to a buildup of dissolved N constituents, and particularly of NO_3 (Wetzel 1983). Chlorophyll a concentrations were lower under ice-cover, supporting this hypothesis. The lower algal biomass under ice-cover was probably a result of reduced light penetration leading to light limitation. There is no obvious explanation for the higher Ni levels in winter.

The pre-development dataset included only one winter (Dec. 1972, Jan. and Feb. 1973) of measurements. This precluded a statistical analysis of changes in seasonal patterns since development, but was sufficient for a qualitative analysis of seasonal differences. This analysis indicated there was no evidence

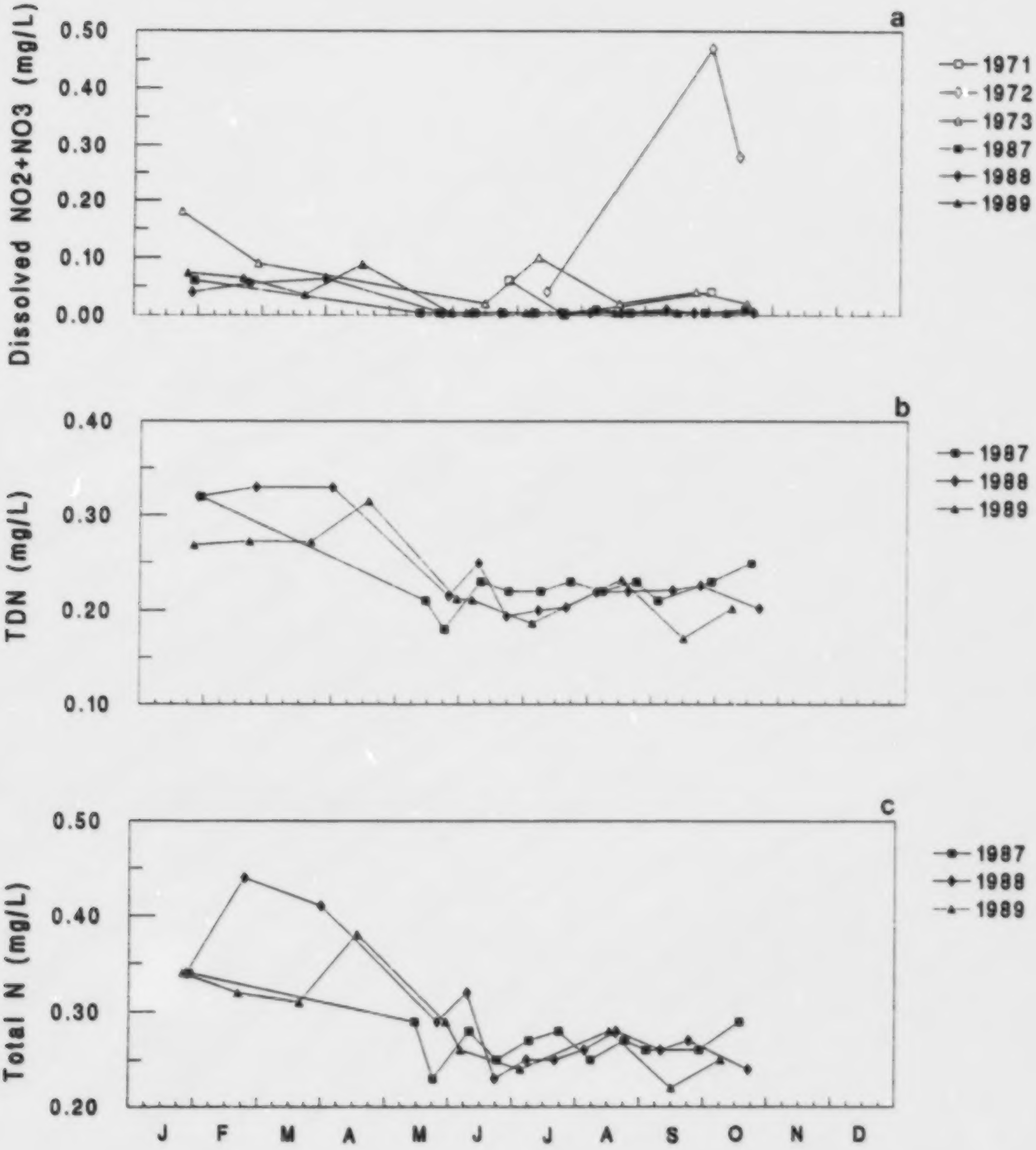


Figure 21. Seasonal water quality trends in the Rat River above Threepoint Lake. a. dissolved NO_2-NO_3 b. TDN c. total N d. chlorophyll a e. dissolved Zn f. total Ni.

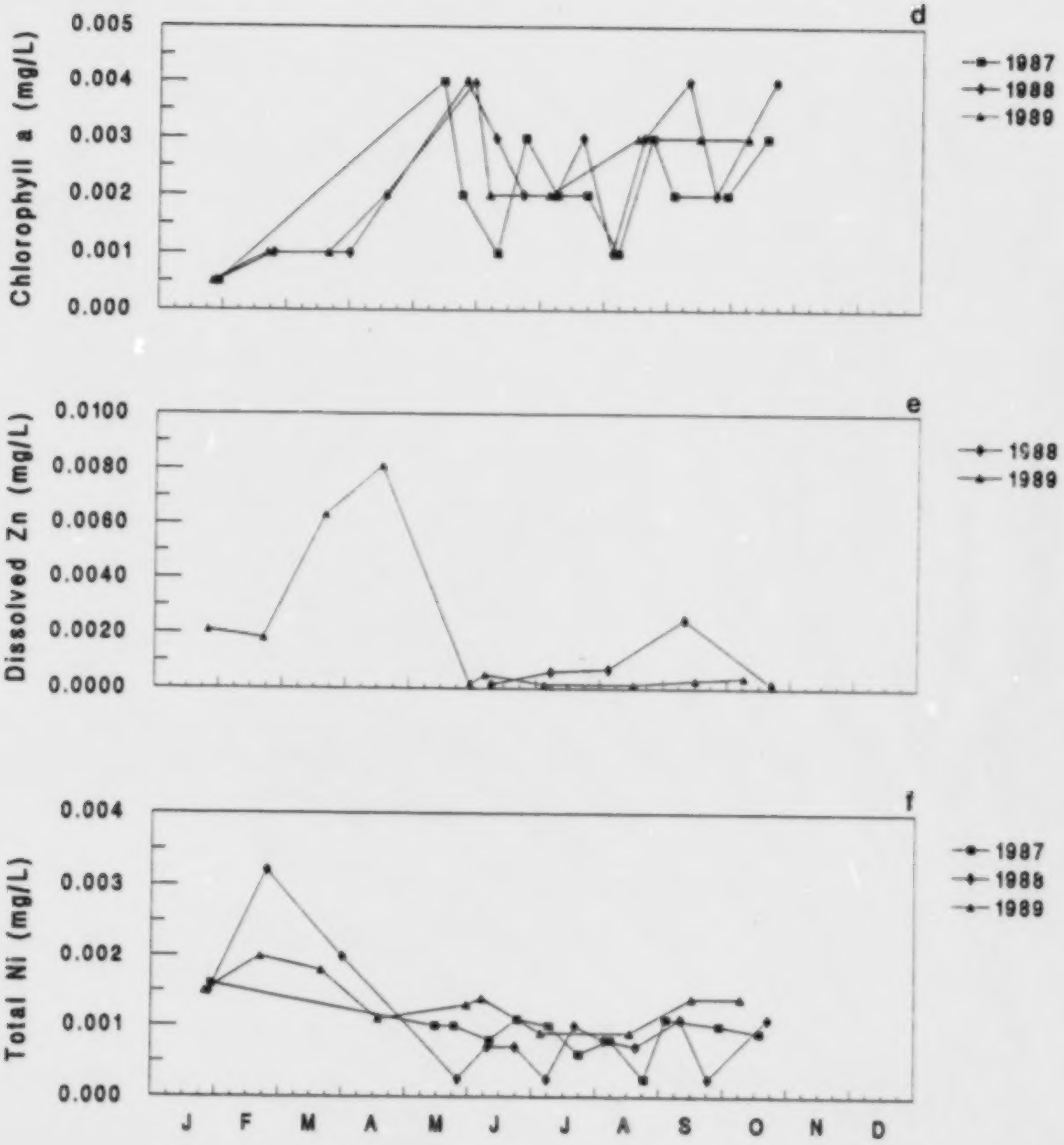


Figure 21. Continued.

of differences between the ice-cover and ice-free seasons in any of the measured variables. However, most of the variables which displayed seasonal trends following diversion were not measured in the pre-diversion studies. Only dissolved NO_2-NO_3 was consistently measured before diversion, and the range of concentrations under ice-cover overlapped the range during the ice-free season. However, the lowest values tended to occur in summer and the highest in winter. The seasonal difference during the FEMP study was just more pronounced than it was prior to diversion. This could be a product of improved analytical or sampling methods rather than representing any effect of CRD.

3.3.1.4 Changes in Water Quality Since Churchill River Diversion

Annual mean conductivity, alkalinity (total), hardness (total), TOC, ortho PO_4, Ca, and reactive Si all decreased by 15 to 60% between the pre-diversion (1972-1973) and post-diversion (1987-1989) sampling periods (Table 16). Significant increases in Na, K, Cl, and F, by 10-80%, also occurred. All of these changes are consistent with the differences in water quality between the Churchill and Rat rivers which existed prior to diversion (Table 17).

The changes in Ca, Cl, Fl, and ortho PO_4 appear to have been a direct result of the replacement of local Rat River flows with water from the Churchill River. However, the changes in conductivity, alkalinity, hardness, Na, and K were smaller than would be expected if the only impact of CRD was substitution of the

Table 16. Evaluation of the significance of changes in selected water quality parameters in the Rat River above Threepoint Lake following Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1972 and 1973. X_{post} denotes the mean for post-development years 1987 through 1989. p is the probability of Student's t-statistic for the comparison.

		pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Tot Alk mg L^{-1}	Tot Hard mg L^{-1}	TOC mg L^{-1}	NO_2NO_3 mg L^{-1}	Ortho PO_4 mg L^{-1}	Tot P mg L^{-1}	Diss Na mg L^{-1}
Rat River	p	0.064	0.002	0.006	0.013	<0.001	0.023	0.049	0.032	0.536	0.010
	X_{pre}	7.9	120.	5.	56.4	61.2	10.7	0.237	0.005	0.023	2.2
	X_{post}	7.66	102.	17.	46.4[1]	45.9[1]	6.38	0.019	0.002[1]	0.026	2.7

	Diss K mg L^{-1}	Diss Ca mg L^{-1}	Diss Cl mg L^{-1}	Diss F mg L^{-1}	Diss SO_4 mg L^{-1}	React Si mg L^{-1}	Diss Fe mg L^{-1}	Diss Mn mg L^{-1}	Diss Cu mg L^{-1}	Diss Pb mg L^{-1}	Diss Zn mg L^{-1}
p	0.004	<0.001	0.049	<0.01	0.630	0.001	0.485	i.d.	1.000	i.d.	0.728
X_{pre}	0.9	18.2	1.0	0.05	2.7	3.6	0.07	<0.01	0.002	<0.001	0.007
X_{post}	1.02	12.3[1]	1.1	0.09	2.5	1.83	0.096	0.003	0.0010	0.0007	0.0014

i.d. Insufficient data.

[1] Discharge-standardized variables.

Table 17. Comparison of water quality in the Rat River before (1972-1973) and after (1987-1989) Churchill River diversion with water quality in the Churchill River near Leaf Rapids (1972-1973). All values are annual means. Units are mg L^{-1} except where noted.

	Rat River		Churchill River
	Pre-Diversion	Post-diversion	
pH (units)	7.9	7.66	7.6
Conductivity ($\mu S\ cm^{-1}$)	120	102	91
Turbidity (NTU)	5	17	5
Alkalinity (Total)	56.4	46.4	40.4
Hardness (Total)	61.2	45.9	41.9
TOC	10.7	6.38	8.7
NO_2NO_3	0.237	0.019	0.157
Ortho PO_4	0.005	0.002	0.003
Total P	0.023	0.026	0.017
Dissolved Na	2.2	2.7	3.0
Dissolved K	0.9	1.02	1.2
Dissolved Ca	18.2	12.3	12.5
Dissolved Cl	1.0	1.1	1.2
Dissolved F	0.005	0.09	0.08
Dissolved SO_4	2.7	2.5	3.6
Reactive Si	3.6	1.83	2.6
Dissolved Fe	0.07	0.096	0.05
Dissolved Mn	<0.01	0.003	<0.01
Dissolved Cu	0.002	0.0010	0.002
Dissolved Pb	<0.001	0.0007	<0.001
Dissolved Zn	0.007	0.0014	0.002

water source. Shoreline erosion along the diversion route may be contributing Na, K, Mg, and carbonates, thereby slightly mitigating the changes in these variables which otherwise would have occurred. Conversely, the decreases in TOC and SO_4 were greater than might be expected. The excess decrease in SO_4 is probably explained by changes in analytical methodology, but the greater decrease in TOC is rather surprising. The flooded vegetation and organic soils in SIL and Notigi Reservoir might rather be expected to contribute organic C to the water column, yet the TOC concentration in the Churchill River appears to decrease after passing through these flooded areas. The cause of this decrease cannot be determined with the available information as it is not known if the loss is occurring in one or both organic carbon fractions (i.e., dissolved or particulate). Possible causes include sedimentation of particulates in Notigi Reservoir or the binding of DOC to other suspended material and subsequent sedimentation. This issue will be examined more extensively in the analysis of water quality data for SIL and Notigi Reservoir being conducted by DFO.

A 92% decline in NO_2-NO_3 also was detected in the analysis (Table 16). This drop is similar to those which occurred in the Burntwood River above Threepoint Lake and in the Footprint River and, therefore, this decrease cannot be attributed to diversion.

Total P concentrations in the Rat River were not lower during the FEMP study than prior to diversion despite lower concentrations in the Churchill River (Table 17). The release of P from flooded soils and vegetation in new reservoirs is well-documented (e.g., Baxter and Glaude 1980). This finding suggests phosphorus continues to be released from flooded terrain as late as 13 years after impoundment and diversion. The lack of data for this site between the start of diversion and the FEMP study prevents examination of the temporal trends in P release which may have been higher in the past.

There also has been an increase in turbidity, by a factor of 3.2, since CRD (Table 16). It is not known if a corresponding increase in suspended sediment (NFR) concentrations has occurred at this site due to the lack of pre-diversion measurements. The increase in turbidity was probably caused by eroding glacial clays in the southern part of Notigi Reservoir or along the Rat River above Threepoint Lake. Erosion is a problem on Southern Indian Lake and in the northern part of Notigi reservoir as well, but this material is thought to be largely retained within Notigi Reservoir (Northwest Hydraulic Consultants 1988). The lack of a correlation between turbidity and discharge at this site ($r^2<0.001$, $p=0.97$, $n=36$) indicates most of the increase in turbidity is due to shoreline erosion in Notigi Reservoir. A positive correlation with discharge would be expected if erosion along the Rat River was the primary cause.

Concentrations of dissolved Fe, Mn, Cu, Pb, and Zn in the Rat River have not been affected by CRD (Table 16). The impact, if any, of diversion on other metal species is unknown due to the lack of pre-development data.

Water quality changes in the Rat River system and in Threepoint Lake have also been examined in studies by Manitoba Natural Resources, Fisheries Branch (Berger and Ramsey 1991, Ramsey et al. 1989). These studies included sample collection in the summers (June-September) of 1986 through 1989 from Rat and Threepoint lakes to document current water quality, and the comparison of these data with published pre-diversion measurements. The water quality changes in the Rat River above Threepoint Lake detected in the present analysis are generally consistent with the changes in Rat and Threepoint lakes following CRD reported in the Fisheries Branch studies. Berger and Ramsey (1991) found lower conductivity, Ca, and reactive Si and higher concentrations of Na and K in both lakes following diversion. Lower Mg concentrations, a variable not found in the pre-development Rat River database, also were found in both lakes following diversion. There was some discrepancy in the impact of CRD on major anions in the two lakes. Concentrations of Cl and SO_4 decreased in Rat Lake but there was no change in either variable in Threepoint Lake. The decrease in Rat Lake was explained by higher concentrations of Cl and SO_4 in the lake than in the Churchill River prior to diversion. The lack of a change in either variable in Threepoint Lake was explained by similar concentrations of both variables in the lake and Churchill River. This finding clarifies the status of the impact of diversion on SO_4 concentrations in the Rat River; a significant decrease must have occurred in the Rat River for there to be no significant change in Threepoint Lake.

The pre-development data sources included in the Berger and Ramsey (1991) study provided better coverage of the nutrient variables in the Rat River system than was possible in the present study; more nutrient variables were measured in Rat Lake both before and after diversion and the NO_3 measurements were not troubled by the analytical problems associated with NO_2-NO_3 in the Rat River. Berger and Ramsey (1991) did not consider Threepoint Lake nutrient changes in detail because the pre-development database for the lake amounted to analyses on a single sample.

Concentrations of the N species in Rat Lake were most affected by CRD. In particular, NO_3 and suspended N concentrations in Rat Lake between 1986 and 1989 were just 23-25% of the 1973 values. Concentrations of DIC, DOC, and suspended C decreased by 26 to 50%, with the lower DOC consistent with the lower post-diversion TOC levels found in the present study. In contrast to the present study, Berger and Ramsey (1991) found suspended P concentrations in Rat Lake dropped 36% on average following diversion while TDP was not affected. Rat Lake P is equally divided between the dissolved and suspended fractions. Therefore, the lower suspended P levels must have caused an approximately 18% decrease in total P, yet this decrease was not evident in the Rat River above Threepoint Lake. This lack of change in the Rat River may be a product of P release from flooded areas between Rat and Threepoint lakes although this possibility remains to be investigated.

3.3.1.5 Adverse Effects

The only adverse effect of CRD on water quality in the Rat River above Threepoint Lake was the 3-fold increase in turbidity. Prior to diversion, mean turbidity met the Canadian drinking water guideline of 5 NTU without treatment. The water should now be treated for removal of turbidity before consumption.

Increased turbidity also has a number of potential ecological implications, although the significance of these impacts remains to be quantified. The higher turbidity might interfere with the ability of sight-feeding fishes to capture food and increased sediment deposition on fish spawning beds can impair whitefish egg survival (Fudge and Bodaly 1984). The elevated turbidity would also be expected to reduce primary productivity in lakes on the Rat River due to reduced light penetration (Hecky and Harper 1974).

3.3.2 Burntwood River at Outlet of Threepoint Lake

3.3.2.1 Current Water Quality

This site was characterized by violations of water quality guidelines similar to those observed in the three rivers which flow into Threepoint Lake. All turbidity measurements exceeded the drinking water guideline, with peak values exceeding the recreational use objective by about a factor of 3 (Table 18). Mean colour and maximum extractable Mn exceeded the drinking water objectives while the mean extractable Fe concentration was about 60% higher than the guidelines for both drinking water and protection of aquatic life. Mean extractable Al exceeded the

Table 18. Summary statistics for physical and chemical water quality variables measured in the Burntwood River at the outlet to Threepoint Lake; September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.5	0.3	8.1	7.0
	Lab		7.59	0.17	7.86	7.28
Conductivity	Field	µS cm^{-1}	102	10	119	80
	Lab		105	5	118	97
Turbidity	Field	NTU	23^{+}	25	$160^{+,++}$	10^{+}
	Lab	NTU	20^{+}	21	$140^{+,++}$	8^{+}
Colour	True	Relative	16^{+}	5	35^{+}	10
TDS		mg L^{-1}	54	4	62	47
NFR	Total	mg L^{-1}	12	21	134	3
	Fixed	mg L^{-1}	11	21	126	2
Alkalinity	Total	mg L^{-1}	47.9	3.2	54.6	42.6
	Bicarbonate	mg L^{-1}	58.4	4.0	66.6	51.9
Hardness	Total	mg L^{-1}	47.3	2.9	53.9	42.0
	Non-carbonate	mg L^{-1}	0.6	1.4	7.0	0
Free CO$_2$		mg L^{-1}	3.09	2.25	9.44	0
Phosphorus	Particulate	mg L^{-1}	0.019	0.018	0.119	<0.006
	Ortho PO4	mg L^{-1}	0.003	0.003	0.012	<0.003
	Total Dissolved	mg L^{-1}	0.010	0.005	0.023	0.003
	Total	mg L^{-1}	0.029	0.018	0.130	0.018
Nitrogen	Particulate	mg L^{-1}	0.054	0.043	0.280	<0.010
	Total Dissolved	mg L^{-1}	0.234	0.039	0.330	0.174
	NO$_2$-NO$_3$	mg L^{-1}	0.016	0.020	0.063	<0.010
	Total NH$_3$	mg L^{-1}	0.012	0.010	0.051	<0.005
	Total	mg L^{-1}	0.29	0.06	0.59	0.23
Carbon	DOC	mg L^{-1}	6.32	0.62	8.50	5.48
	POC	mg L^{-1}	0.380	0.317	2.100	0.150
	TOC	mg L^{-1}	6.70	0.74	8.80	5.77
	DIC	mg L^{-1}	11.3	0.7	13.1	9.8
Chlorophyll-a		mg L^{-1}	0.003	0.001	0.007	<0.001
Ca	Dissolved	mg L^{-1}	12.7	0.9	14.7	10.9
Mg	Dissolved	mg L^{-1}	3.8	0.3	4.6	3.4
K	Dissolved	mg L^{-1}	1.03	0.07	1.24	0.94
Na	Dissolved	mg L^{-1}	2.7	0.3	3.5	2.2

Table 18. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	1.91	0.71	3.36	0.76
Cl	Dissolved	mg L^{-1}	1.1	0.2	1.7	0.8
F	Dissolved	mg L^{-1}	0.09	0.01	0.13	0.07
SO$_4$	Dissolved	mg L^{-1}	2.4	0.6	4.3	0.8
Al	Extractable	mg L^{-1}	0.377^{+++}	0.209	1.360^{+++}	<0.100
	Dissolved	mg L^{-1}	<0.100	0.038	0.179	<0.100
As	Dissolved	mg L^{-1}	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L^{-1}	<0.08	0.02	0.13	<0.08
Cd	Total	mg L^{-1}	0.0001	0.0001	0.0006^{+++}	<0.0001
	Dissolved	mg L^{-1}	<0.0001		0.0002	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0003	0.0020	<0.0005
Cu	Total	mg L^{-1}	0.0044^{+++}	0.0059	0.0264^{+++}	0.0005
	Dissolved	mg L^{-1}	0.0010	0.0004	0.0017	0.0005
Fe	Extractable	mg L^{-1}	0.477$^{+,+++}$	0.303	2.010$^{+,+++}$	0.135
	Dissolved	mg L^{-1}	0.103	0.109	0.627	0.020
Pb	Total	mg L^{-1}	0.0008	0.0009	0.0046^{+++}	<0.0007
	Dissolved	mg L^{-1}	0.0009	0.0015	0.0050	<0.0007
Mn	Extractable	mg L^{-1}	0.020	0.028	0.179^{+}	0.006
	Dissolved	mg L^{-1}	0.004	0.004	0.017	<0.002
Ni	Total	mg L^{-1}	0.0013	0.0009	0.0059	<0.0005
	Dissolved	mg L^{-1}	0.0005	0.0004	0.0015	<0.0005
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0005	<0.0001
V	Total	mg L^{-1}	0.0010	0.0006	0.0039	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0001	0.0006	<0.0005
Zn	Total	mg L^{-1}	0.0040	0.0040	0.0182	0.0007
	Dissolved	mg L^{-1}	0.0011	0.0015	0.0056	<0.0003
Discharge	Daily	m^3 sec^{-1}	832	105	1020	670

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recreational use.

$^{+++}$ Exceeds limit for protection of aquatic life.

aquatic life guideline by a factor of 3.7 on average and by as much as a factor of 13.6 at peak concentrations. Total Cu exceeded the aquatic life guideline by a factor of 2 on average and by as much as a factor of 13. Peak total Cd and total Pb also exceeded the respective aquatic life guidelines.

3.3.2.2 Water Quality - Water Discharge Relationships

All significant correlations between water quality and discharge were positive at this site. These correlates included conductivity (lab), TDS, alkalinity (total and bicarbonate), total hardness, and extractable Al (Figure 22). The slopes and intercepts of discharge relationships for alkalinity, TDS and hardness at this site were the same as in the Rat River above Threepoint Lake as was the generally weak character of the correlations. Only 12-21% of the variance in the dependent variables was accounted for by the regression on discharge. There was no resemblance between the water chemistry - discharge relationships in the unregulated inflows (Burntwood and Footprint rivers) to those at the outflow (Figures 8, 9, and 22). Again, the discharge correlations for this site are of limited use for water quality management or prediction due to the weakness of the relationships and the lack of a correlation for any variable responsible for an adverse effect at this site or downstream.

No examination of the pre-development relationships between water quality and discharge is possible for this site due to the lack of pre-development discharge and water quality data. These

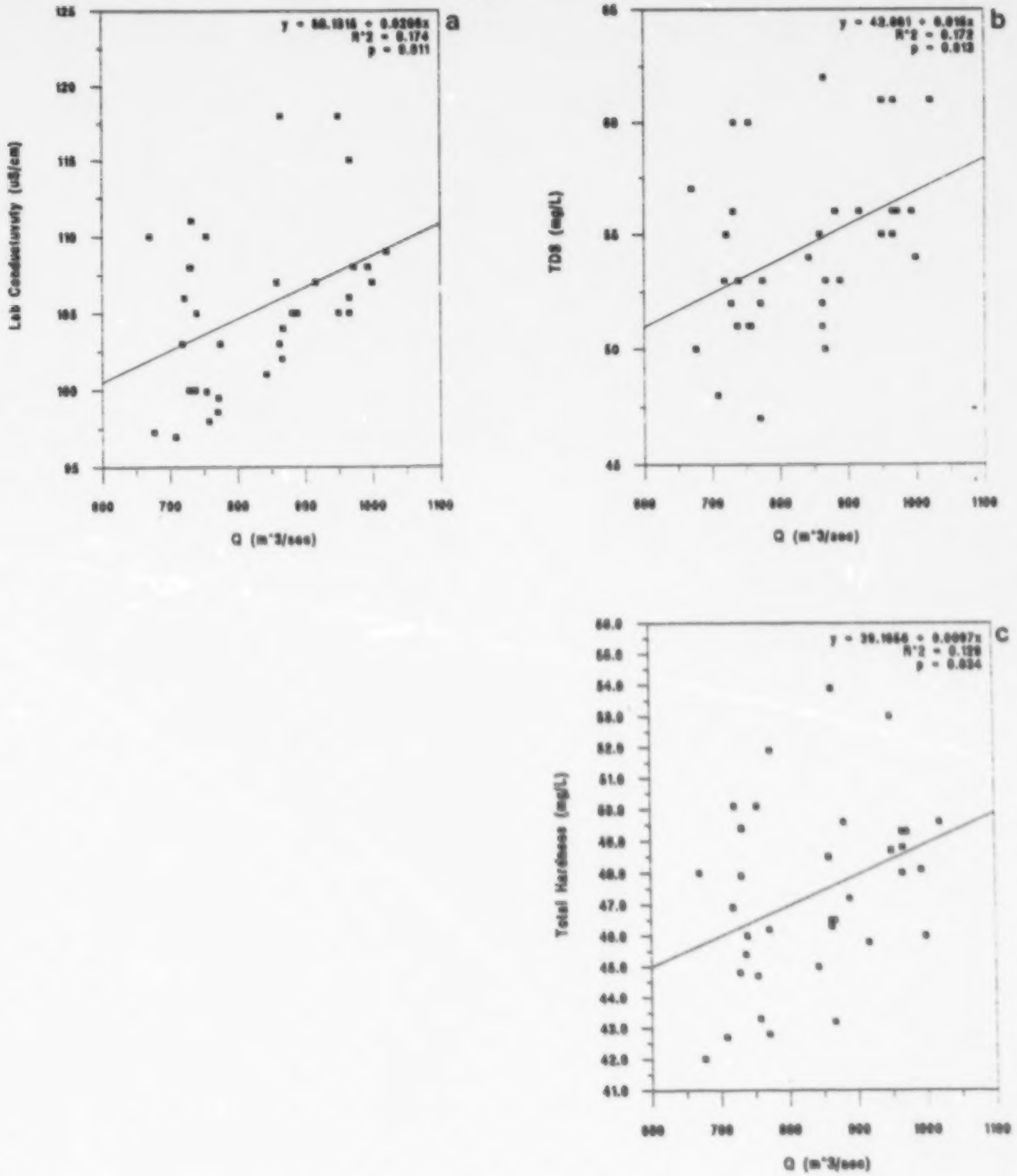


Figure 22. Water chemistry - water discharge relationships in the Burntwood River at the outlet of Threepoint Lake, 1987-1989. a. lab conductivity b. TDS c. total hardness d. total alkalinity e. bicarbonate alkalinity f. extractable Al.

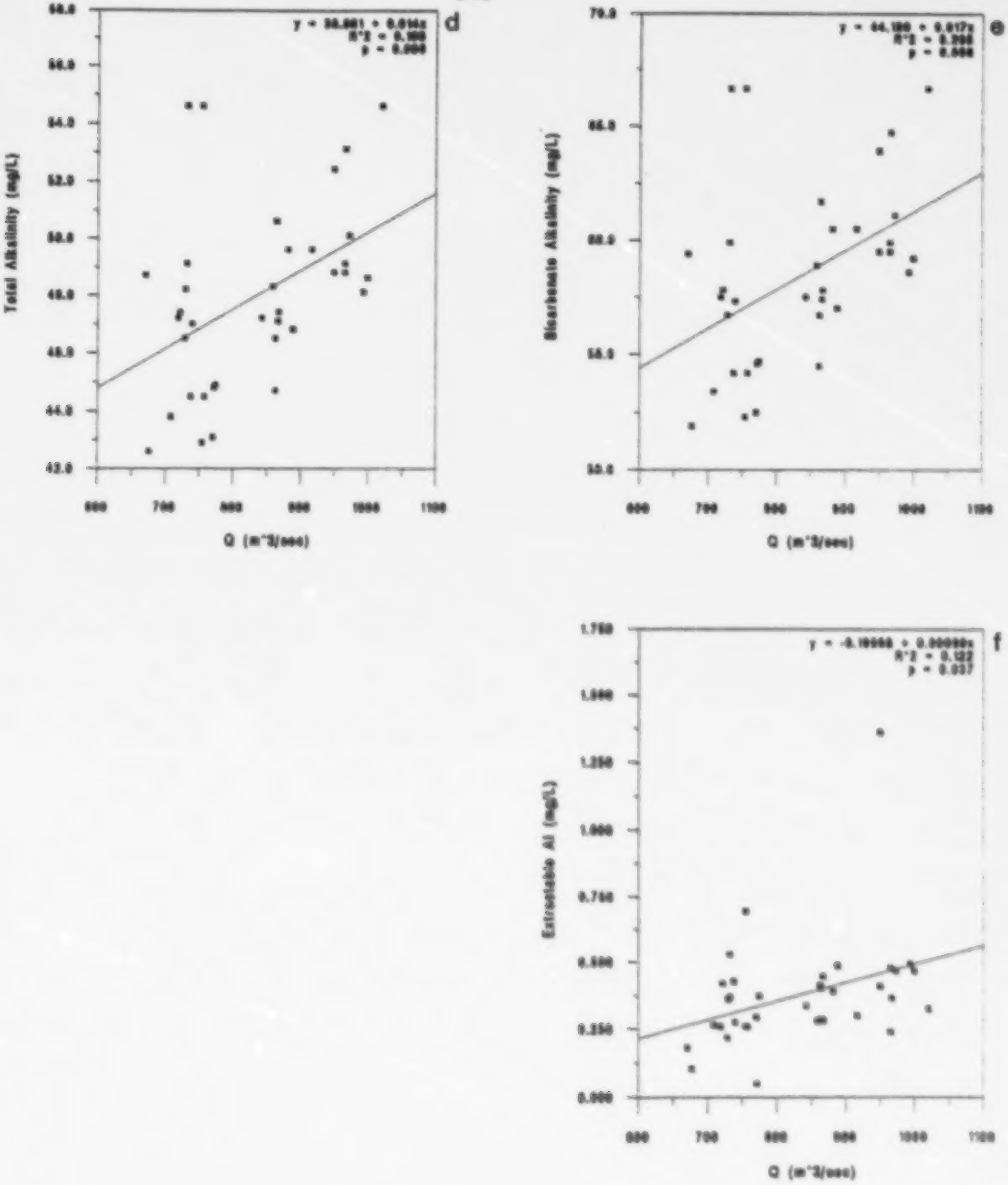


Figure 22. Continued.

relationships probably were most similar to those in the Burntwood River above Threepoint lake as this was the major inflow to the lake prior to diversion.

3.3.2.3 Seasonal Differences

Few differences in water quality were evident between the ice-free and ice-cover seasons during the 1987-1989 FEMP study period. As in the Rat River at the inlet to Threepoint Lake, TDN and dissolved NO_2-NO_3 concentrations were higher under ice-cover, probably due to the same cause as in the Rat River (Figure 23). Dissolved Zn concentrations also were much higher under ice-cover. The highest dissolved Zn concentrations occurred under ice cover in all three major inflows to Threepoint Lake as well, although the seasonality was not as distinct as at the outlet. The cause of this winter increase is unknown. The dynamics of Zn in freshwaters has been little studied (Wetzel 1983).

3.3.3 Nelson House Bacterial Testing

Geometric mean fecal coliform bacteria counts at the eight survey sites generally ranged between 1 and 6 bacteria per 100 mL for the seasonal study (Figure 24). The only exception was School Bay, where the geometric mean count was 56 per 100 mL. Fecal coliform counts in water samples from Industrial Bay, Church Bay (East Ramp), Government Dock, Inspiration Point, New Development Area (Across from Cemetery), and Metis Beach were consistently below 200 per 100 mL. Counts in 4 of the 5 samples from Nursing

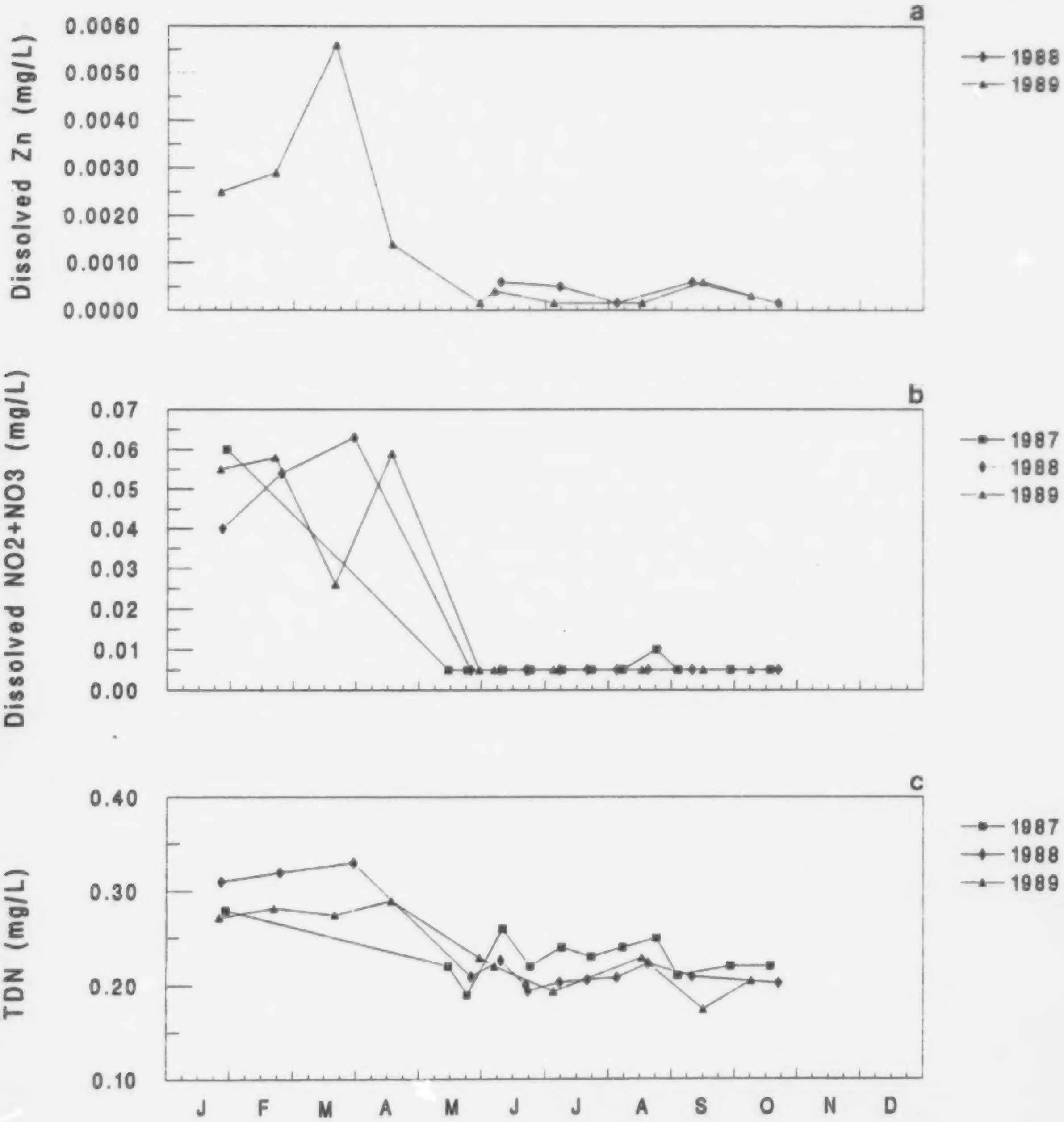
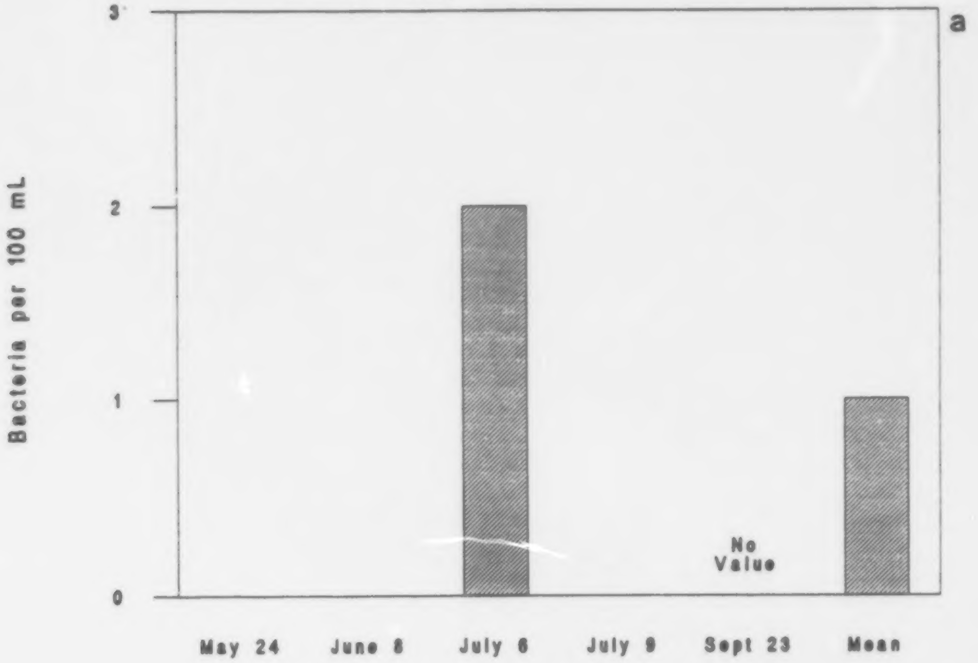


Figure 23. Seasonal water quality trends in the Burntwood River at the outlet of Threepoint Lake. a. dissolved Zn b. dissolved NO_2-NO_3 c. TDN.



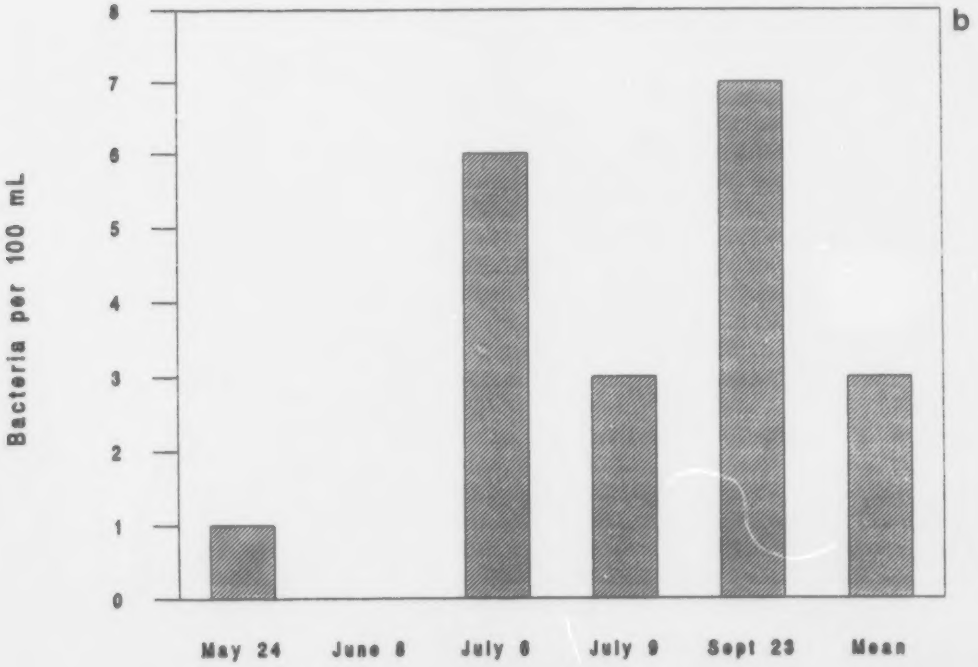
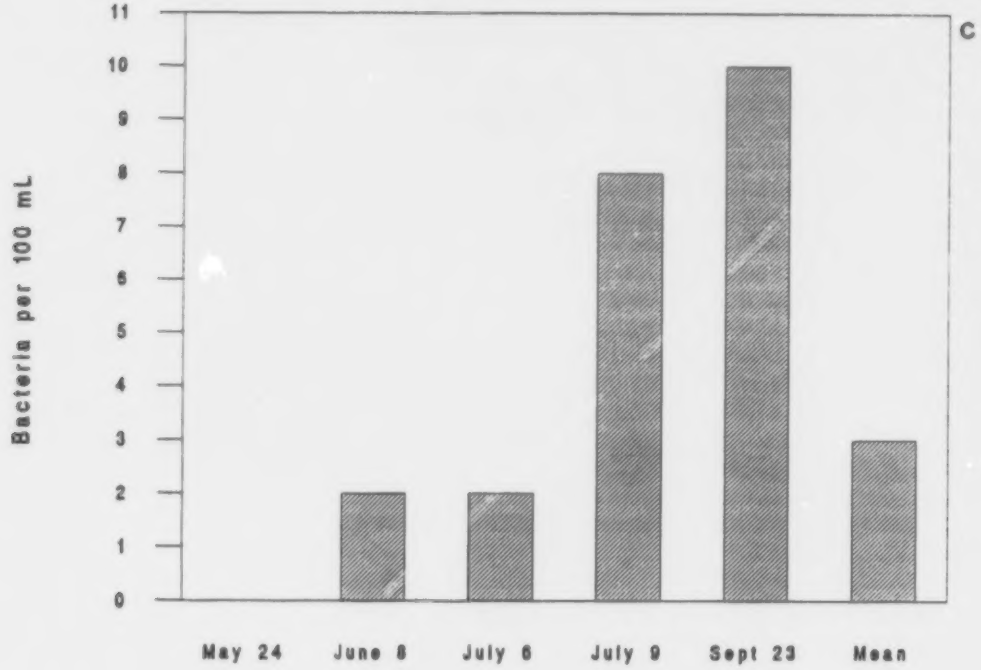
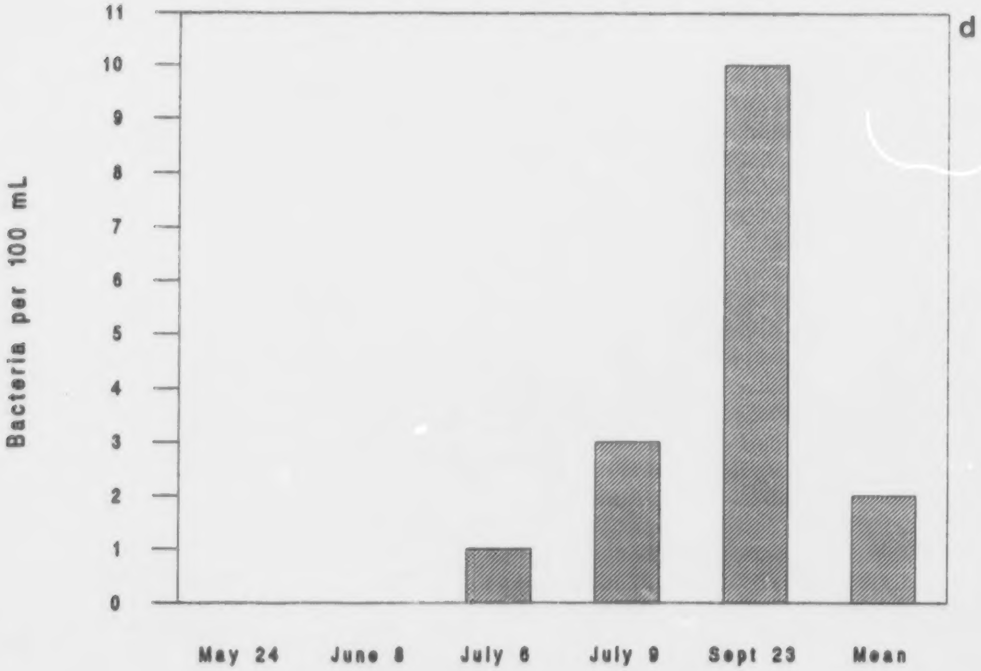


Figure 24. Fecal coliform bacteria counts from the seasonal study in the vicinity of Nelson House. Mean is the geometric mean. Dotted line indicates Canadian Water Quality Guideline. a. Church Bay (East Ramp) b. Government Dock c. Industrial Bay d. Inspiration Point e. Metis Beach f. New Development Area (Across from Cemetery) g. Nursing Station Bay h. School Bay.





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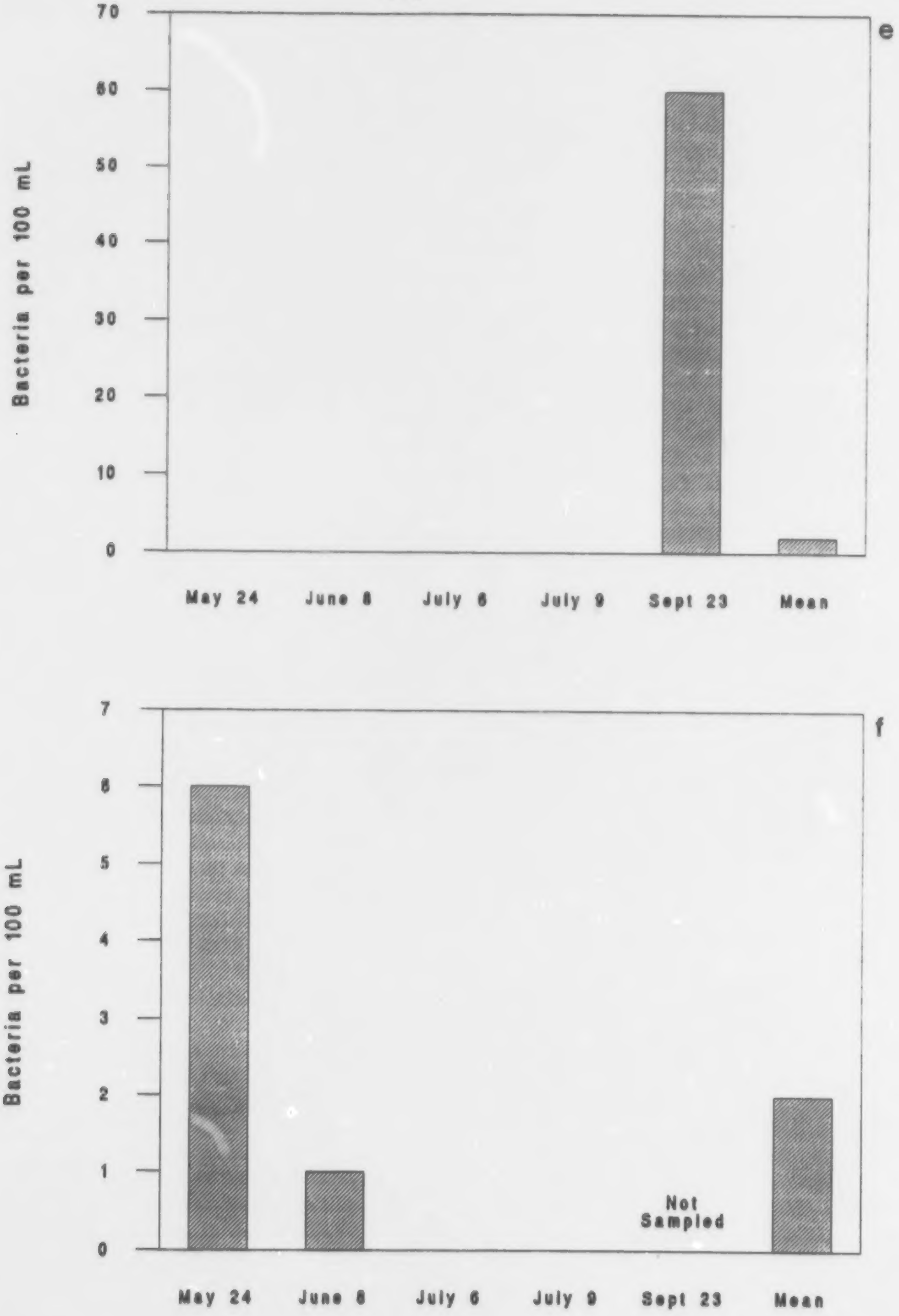

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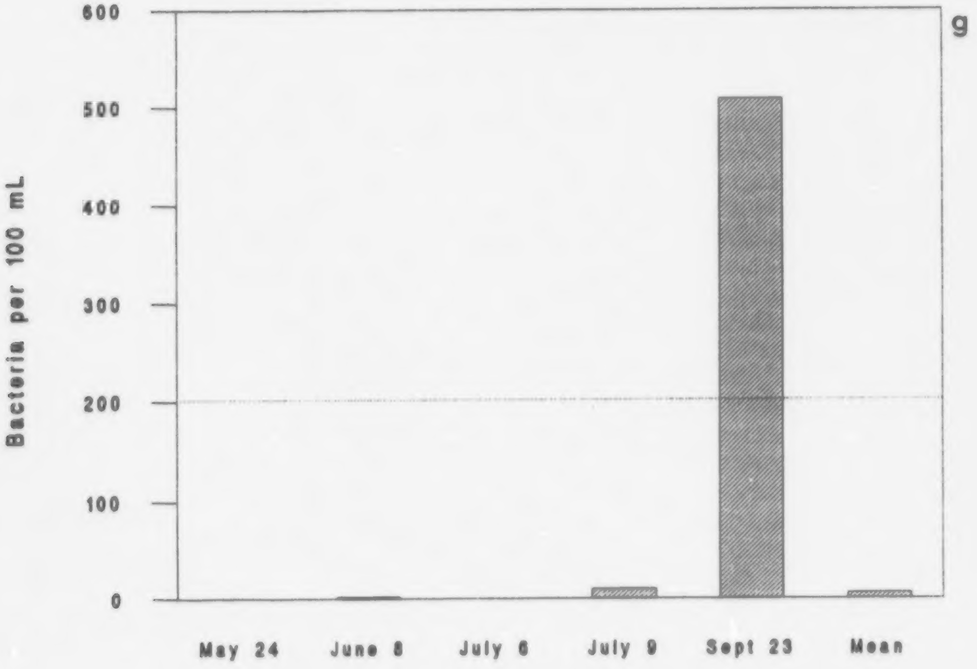


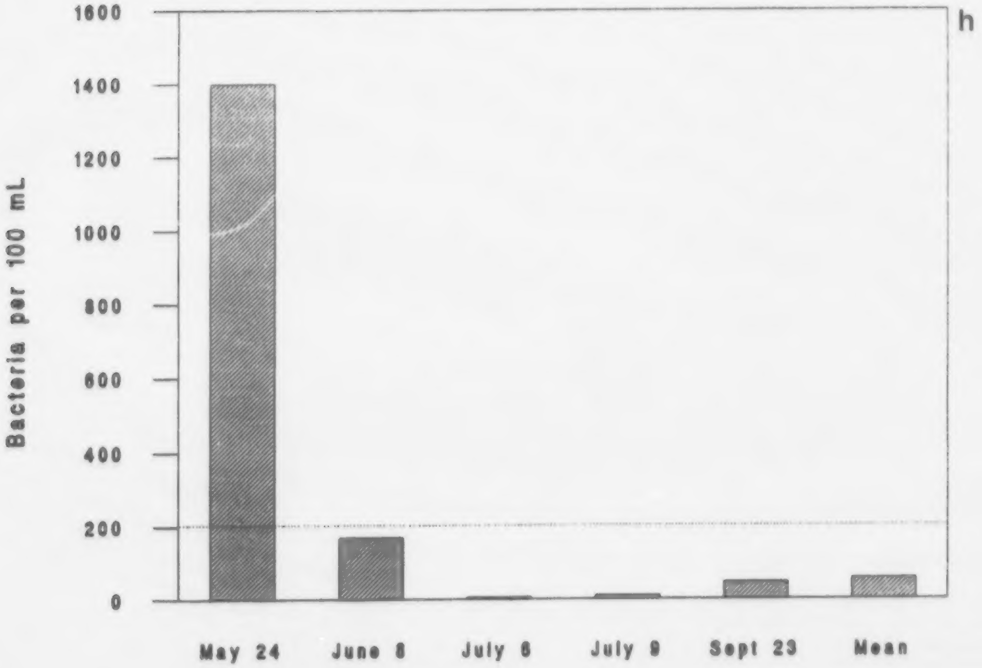


Figure 24. Continued.

Station Bay were well below 200 bacteria per 100 mL while a count of 508 per 100 mL was found on September 23, 1987.

Intensive studies of bacterial water quality in the vicinity of the sewage treatment plant outfall were undertaken on August 12 and 25, 1987. Fecal coliform counts in 16 of the 19 samples were below 50 per 100 mL (Figure 25). Counts in the other 3 samples ranged between 530 and 4100 per 100 mL. The geometric mean of the sample counts was 38 per 100 mL with 16% of the samples exceeding 400 counts per 100 mL. The few high coliform counts were transient. At sites 3, 7, and 8, respectively, counts were 4100, 43, and 530 per 100 mL on August 12, compared with 7, 1910, and 0 on August 25.

Bacteriological water quality at the Nelson House study sites was generally acceptable for recreational water uses, as indicated by the Health and Welfare Canada Recreational Water Quality Guideline of 200 bacteria per 100 mL. The intensive study demonstrated this criterion was met even in the area of the sewage treatment plant outfall.

3.4 Burntwood River at Thompson

The Burntwood River at Thompson is the only site in the FEMP study which was sampled before, during, and immediately after construction of an hydroelectric project in addition to the current study. This site also has the best baseline data record of any in the study, sampled continuously from 1967 to 1974. Only 10 variables were common to the four periods, however, thus seriously limiting the nature of the water quality changes which can be

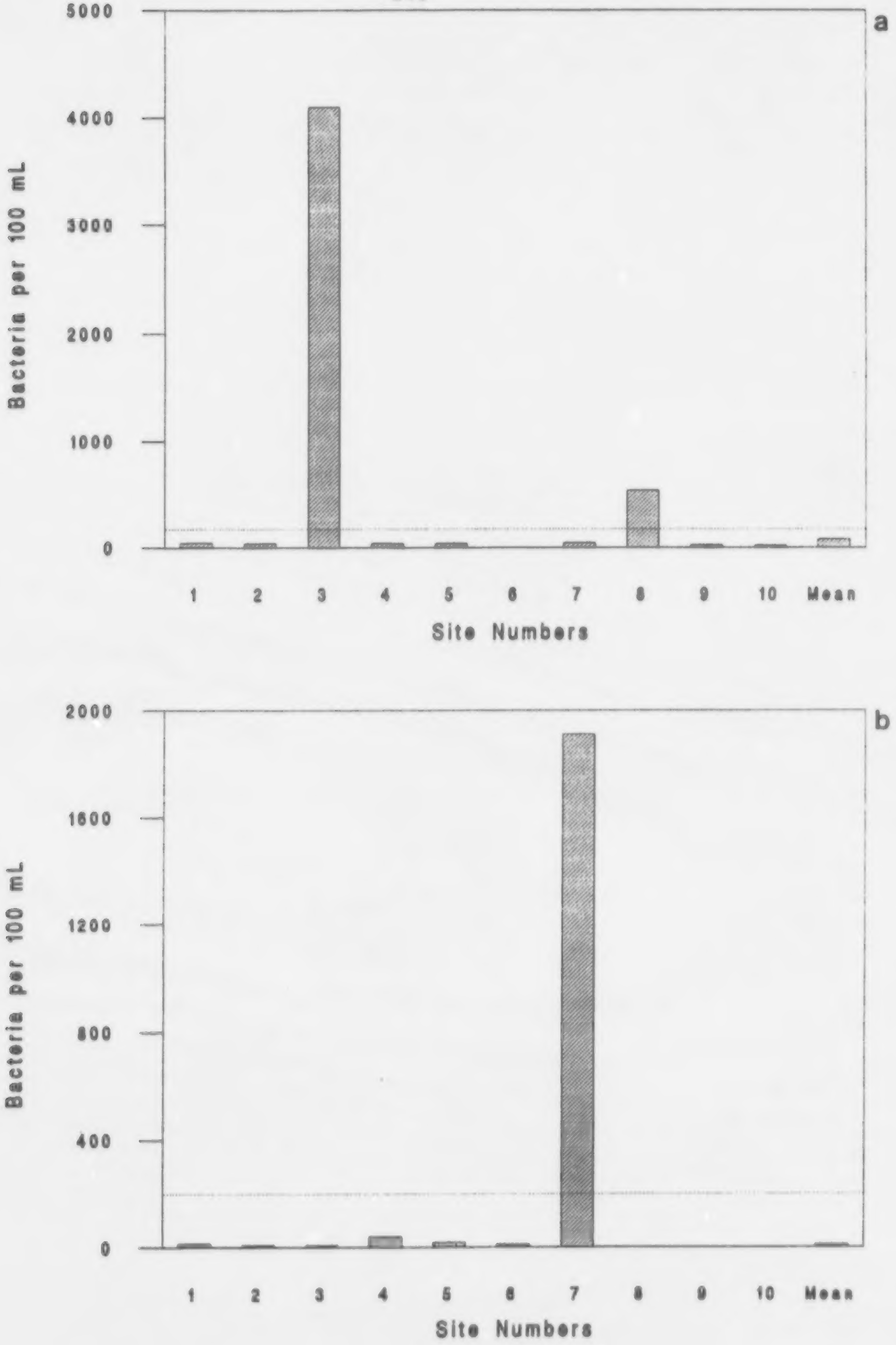


Figure 25. Fecal coliform bacteria counts from the intensive study in the vicinity of the Nelson House Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 12 August 1987 b. Sampled 25 August 1987. See Appendix 1 for site locations and descriptions.

conclusively stated. A number of variables were added to the analytical program during the construction period, and changes have occurred in several of these since diversion. The lack of measurements in the pre-diversion period raises some question about the directness of any relationship between these changes and CRD. In some cases there may be indirect evidence that the change is applicable to the pre-development period, but in other instances, there is no way to be certain.

3.4.1 Current Water Quality

Water quality in the Burntwood River near Thompson had similar characteristics to the other sites in the local (Burntwood, Footprint, and Aiken rivers) and Churchill River drainages. Levels of most parameters were well within Canadian guidelines, and those which exceeded the guidelines were the same as in the local drainages. All turbidity measurements exceeded the drinking water guideline, by about a factor of 6 on average (Table 19). The peak level measured in the field also exceeded the guideline for recreational uses by almost 30%. Colour exceeded the drinking water objective by about 40% on average and by as much as 100%. Extractable Fe exceeded the aquatic life and drinking water objectives by a factor of 2 on average and by as much as a factor of 3. All measurements of extractable Al exceeded the aquatic life guideline; by a factor of 4.9 on average and by as much as a factor of 8.3. The mean total Cu concentration was a factor 2 higher than the aquatic life guideline on average and as much as a factor 12

Table 19. Summary statistics for physical and chemical water quality in the Burntwood River at Thompson, January, 1987, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.6	0.3	8.2	7.1
	Lab		7.68	0.24	7.96	6.80
Conductivity	Field	μS cm^{-1}	111	10	142	90
	Lab		116	6	133	104
Turbidity	Field	NTU	31^{+}	9	$65^{+,++}$	18^{+}
	Lab		27^{+}	6	37^{+}	15^{+}
Colour	True	Relative	21^{+}	5	30^{+}	10
TDS		mg L^{-1}	60	4	68	54
NFR	Total	mg L^{-1}	17	5	32	7
	Fixed	mg L^{-1}	15	5	29	6
Alkalinity	Total	mg L^{-1}	53.1	3.1	59.9	46.4
	Bicarbonate	mg L^{-1}	64.7	3.8	73.0	56.6
Hardness	Total	mg L^{-1}	52.8	3.2	62.7	47.6
	Non-carbonate	mg L^{-1}	0.6	1.2	6.5	0
Free CO_2		mg L^{-1}	2.86	2.01	7.97	0
Phosphorus	Particulate	mg L^{-1}	0.023	0.008	0.042	<0.006
	Ortho PO_4	mg L^{-1}	0.003	0.005	0.027	<0.003
	Total Dissolved	mg L^{-1}	0.012	0.005	0.026	0.003
	Total	mg L^{-1}	0.036	0.007	0.056	0.025
Nitrogen	Particulate	mg L^{-1}	0.046	0.020	0.090	<0.010
	Total Dissolved	mg L^{-1}	0.249	0.047	0.436	0.190
	NO_2-NO_3	mg L^{-1}	0.022	0.038	0.210	<0.010
	Total NH_3	mg L^{-1}	0.015	0.011	0.060	<0.005
	Total	mg L^{-1}	0.30	0.04	0.48	0.23
Carbon	DOC	mg L^{-1}	6.49	0.55	7.60	5.58
	POC	mg L^{-1}	0.457	0.163	0.940	0.224
	TOC	mg L^{-1}	6.95	0.64	8.54	6.02
	DIC	mg L^{-1}	12.4	1.2	14.4	6.7
Chlorophyll-a		mg L^{-1}	0.002	0.001	0.005	<0.001
Ca	Dissolved	mg L^{-1}	14.6	1.0	17.7	12.9
Mg	Dissolved	mg L^{-1}	4.0	0.2	4.6	3.6
K	Dissolved	mg L^{-1}	1.05	0.07	1.28	0.95
Na	Dissolved	mg L^{-1}	2.7	0.3	3.4	2.2

Table 19. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	2.01	0.71	3.32	0.78
Cl	Dissolved	mg L^{-1}	1.2	0.2	1.8	0.8
F	Dissolved	mg L^{-1}	0.09	0.01	0.14	0.07
SO_4	Dissolved	mg L^{-1}	2.8	0.7	5.1	1.7
Al	Extractable	mg L^{-1}	0.491^{+++}	0.154	0.826^{+++}	0.230^{+++}
	Dissolved	mg L^{-1}	<0.100	0.043	0.179	<0.100
As	Dissolved	mg L^{-1}	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L^{-1}	<0.08		<0.08	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0002	<0.0001
	Dissolved	mg L^{-1}	0.0001	0.0001	0.0003	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0002	0.0011	<0.0005
Cu	Total	mg L^{-1}	0.0040^{+++}	0.0061	0.0240^{+++}	0.0009
	Dissolved	mg L^{-1}	0.0014	0.0006	0.0034	0.0008
Fe	Extractable	mg L^{-1}	0.624$^{+,+++}$	0.189	0.931$^{+,+++}$	0.256
	Dissolved	mg L^{-1}	0.076	0.041	0.156	0.014
Pb	Total	mg L^{-1}	0.0009	0.0006	0.0027^{+++}	<0.0007
	Dissolved	mg L^{-1}	0.0012	0.0016	0.0057	<0.0007
Mn	Extractable	mg L^{-1}	0.021	0.005	0.030	0.005
	Dissolved	mg L^{-1}	0.004	0.003	0.012	<0.002
Ni	Total	mg L^{-1}	0.0018	0.0014	0.0089	0.0007
	Dissolved	mg L^{-1}	0.0007	0.0004	0.0015	<0.0005
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0003	<0.0001
V	Total	mg L^{-1}	0.0013	0.0005	0.0025	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0002	0.0007	<0.0005
Zn	Total	mg L^{-1}	0.0035	0.0030	0.0172	0.0012
	Dissolved	mg L^{-1}	0.0012	0.0013	0.0051	<0.0003
Discharge	Daily	m^3 sec^{-1}	874	102	1080	718

+ Exceeds limit for drinking water.

++ Exceeds limit for recreational use.

+++ Exceeds limit for protection of aquatic life.

higher. The maximum total Pb levels measured at this site also exceeded the aquatic life guideline while the mean concentration was within 10% of the guideline level.

3.4.2 Water Quality - Water Discharge Relationships

3.4.2.1 Pre-development, 1967-1974

Most correlations between water quality and discharge were negative during the pre-development period. The negative correlates included conductivity (lab), total alkalinity, total hardness, dissolved Na, and reactive Si (Figure 26). The only positive correlate was turbidity (lab). Most of the regressions on discharge accounted for 40 to 50% of the variance in the dependent variables, although the 22% of the variance in reactive Si was attributable to discharge.

3.4.2.2 Diversion Construction, 1974-1976

All significant correlations were negative during the construction period (Figure 27), during which Rat River flows at the outlet of Notigi Lake were completely cut off. All of the discharge-correlated variables in this period (conductivity (lab), total alkalinity, total hardness, and dissolved Na) were common to the pre-development period as well. The y-intercepts of these relationships varied little between the two periods but the slopes were generally steeper during the construction period. The goodness of fit of the correlations for the construction period were as high or higher than prior to development, with the regressions

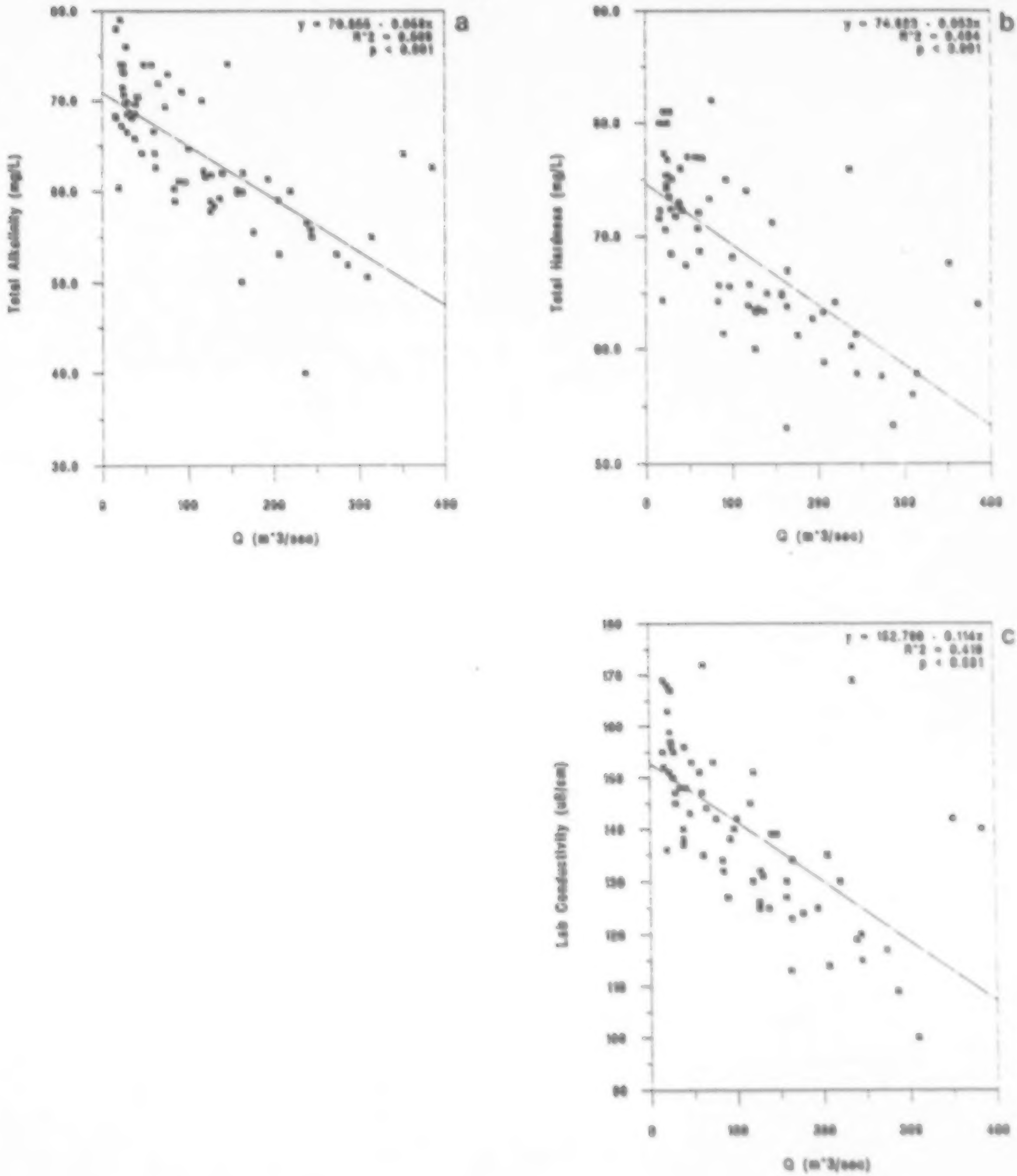
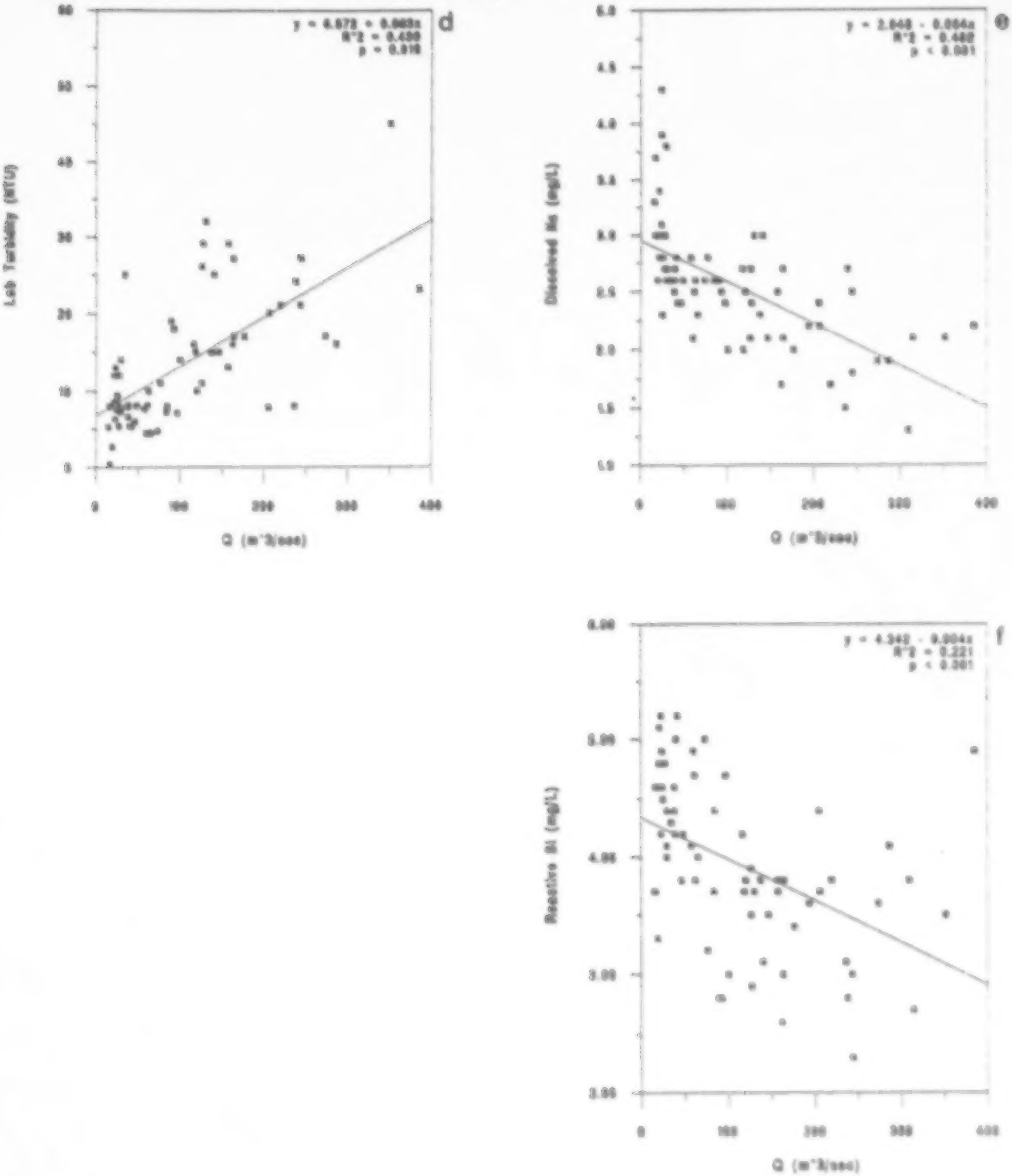


Figure 26. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the pre-development period; from February 1967 to May 1974. a. total alkalinity b. total hardness c. lab conductivity d. lab turbidity e. dissolved Na f. reactive Si.



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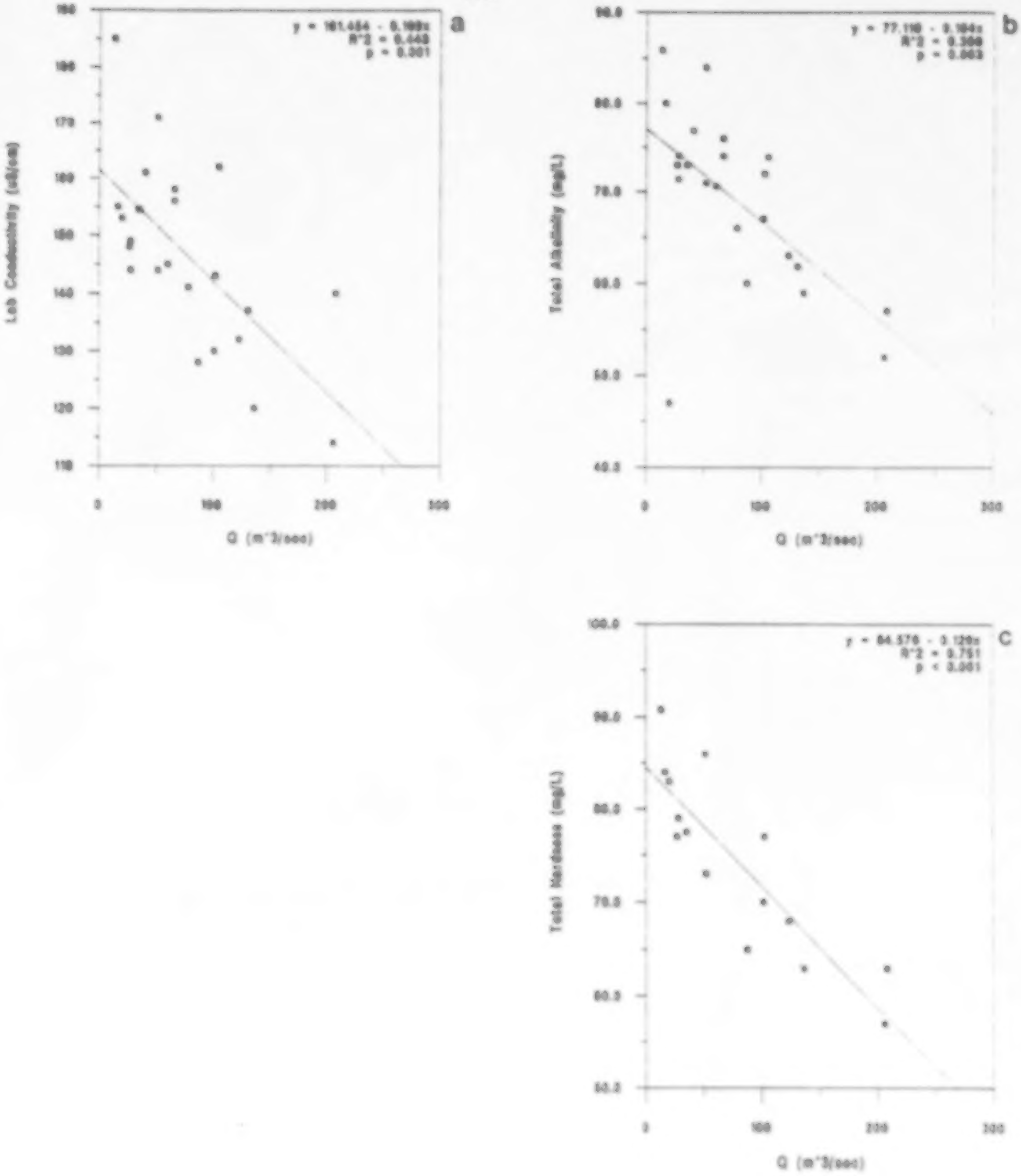


Figure 27. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the construction period; from June 1974 to August 1976. a. lab conductivity b. total alkalinity c. total hardness d. dissolved Mg e. dissolved Na.

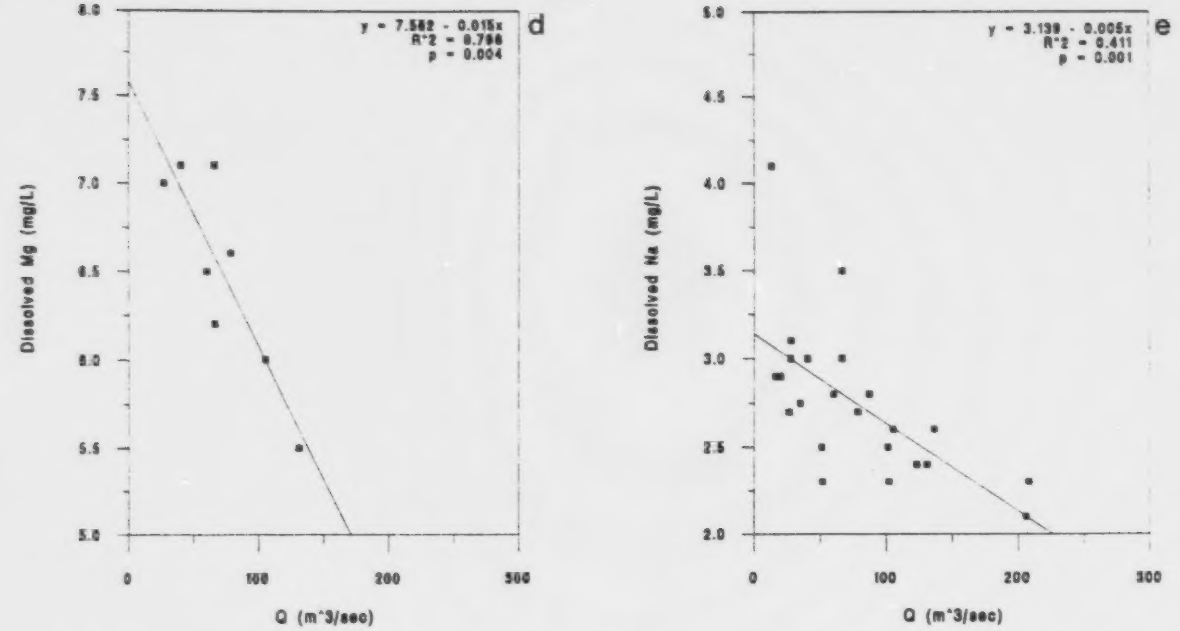


Figure 27. Continued.

accounting for 36 to 77% of the variance. Turbidity and reactive Si, which were correlated with discharge in the pre-development period, were not correlated with discharge during the construction period.

Dissolved Mg was not measured in the pre-development period. Its correlation with discharge is consistent with that of hardness, of which Mg is a major component (APHA 1985).

3.4.2.3 Immediate Post-diversion, 1976-1978

Again, all significant correlations were negative during this period (Figure 28). The correlates included total alkalinity and total hardness, as in both the pre-development and construction periods, reactive Si, as in the pre-development period, and pH. Turbidity, conductivity, and dissolved Na and Mg were not correlated with discharge in this period. The slopes of the alkalinity, hardness, and reactive Si relationships were substantially lower than in the preceding periods, although the y-intercepts remained similar. All four regressions accounted for 30 to 40% of the variance in the dependent variables, similar to those in the pre-development period.

3.4.2.4 Current, 1987-1989

All significant correlations with discharge were positive at this site (Figure 29), as found upstream at the outlet of Threepoint Lake. Correlates included turbidity (lab), NFR (total and fixed), alkalinity (total and bicarbonate), particulate and

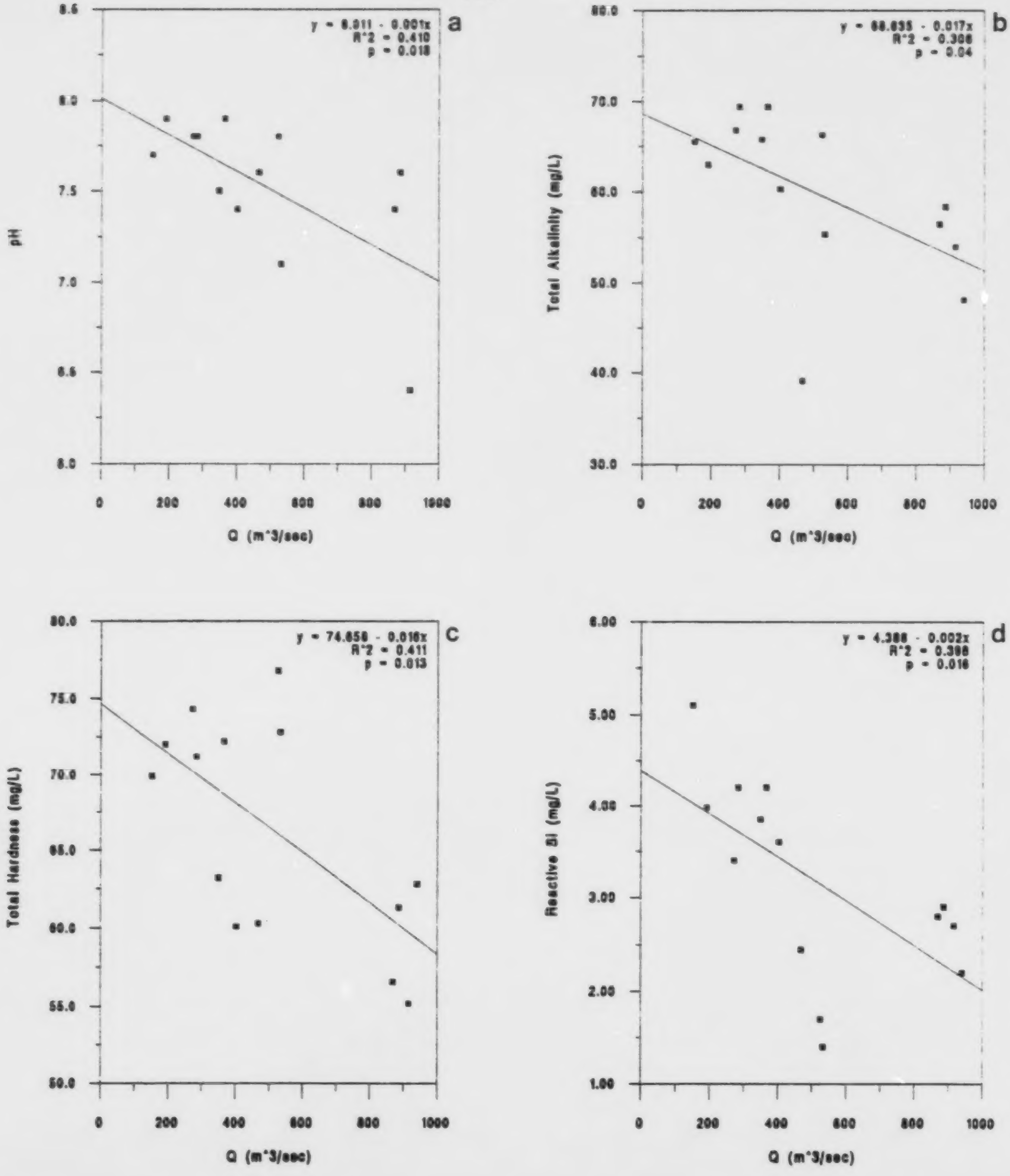


Figure 28. Water chemistry - water discharge relationships in the Burntwood River at Thompson for the immediate post-diversion period; from September 1976 to December 1977. a. pH b. total alkalinity c. total hardness d. reactive Si.

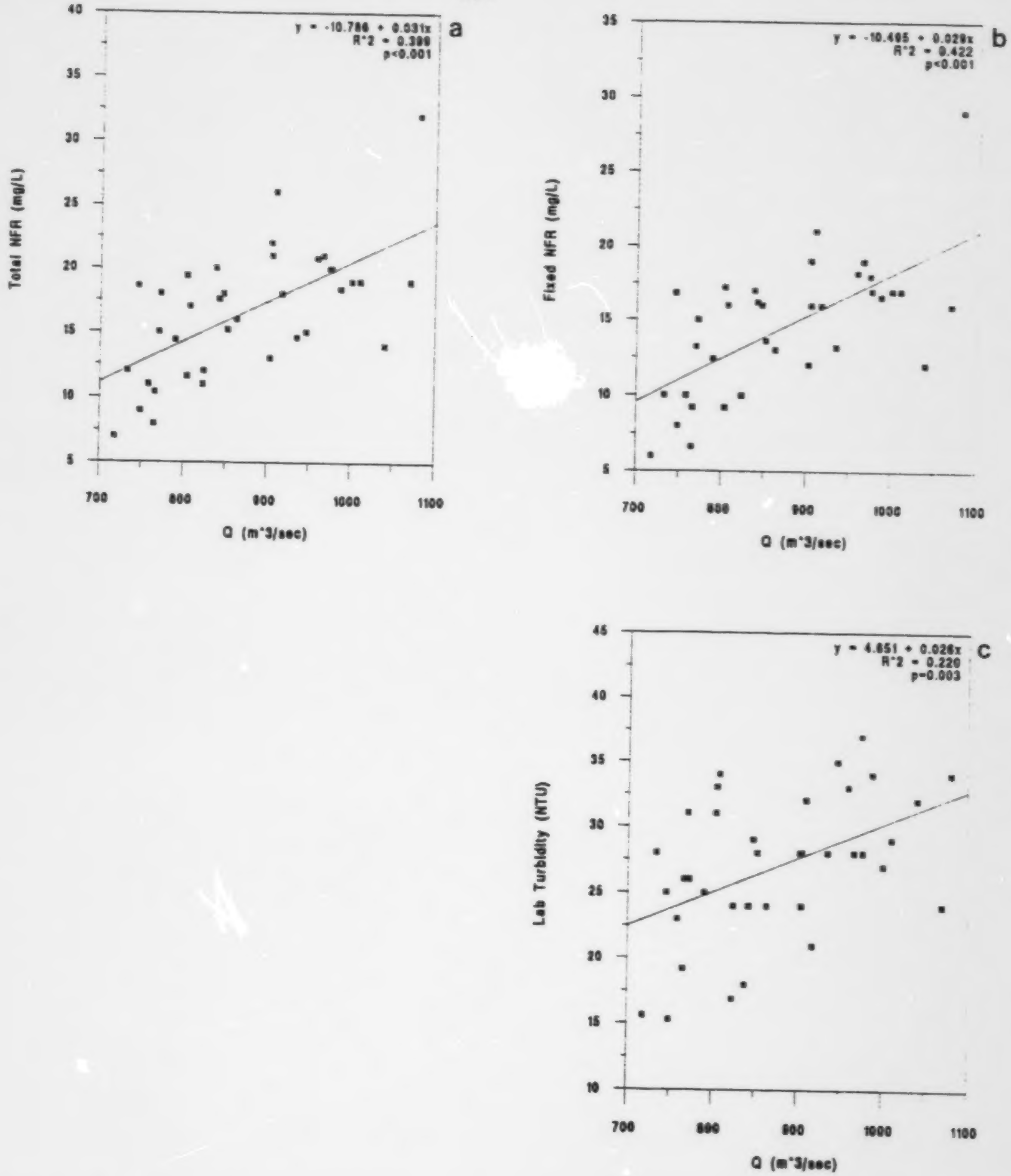
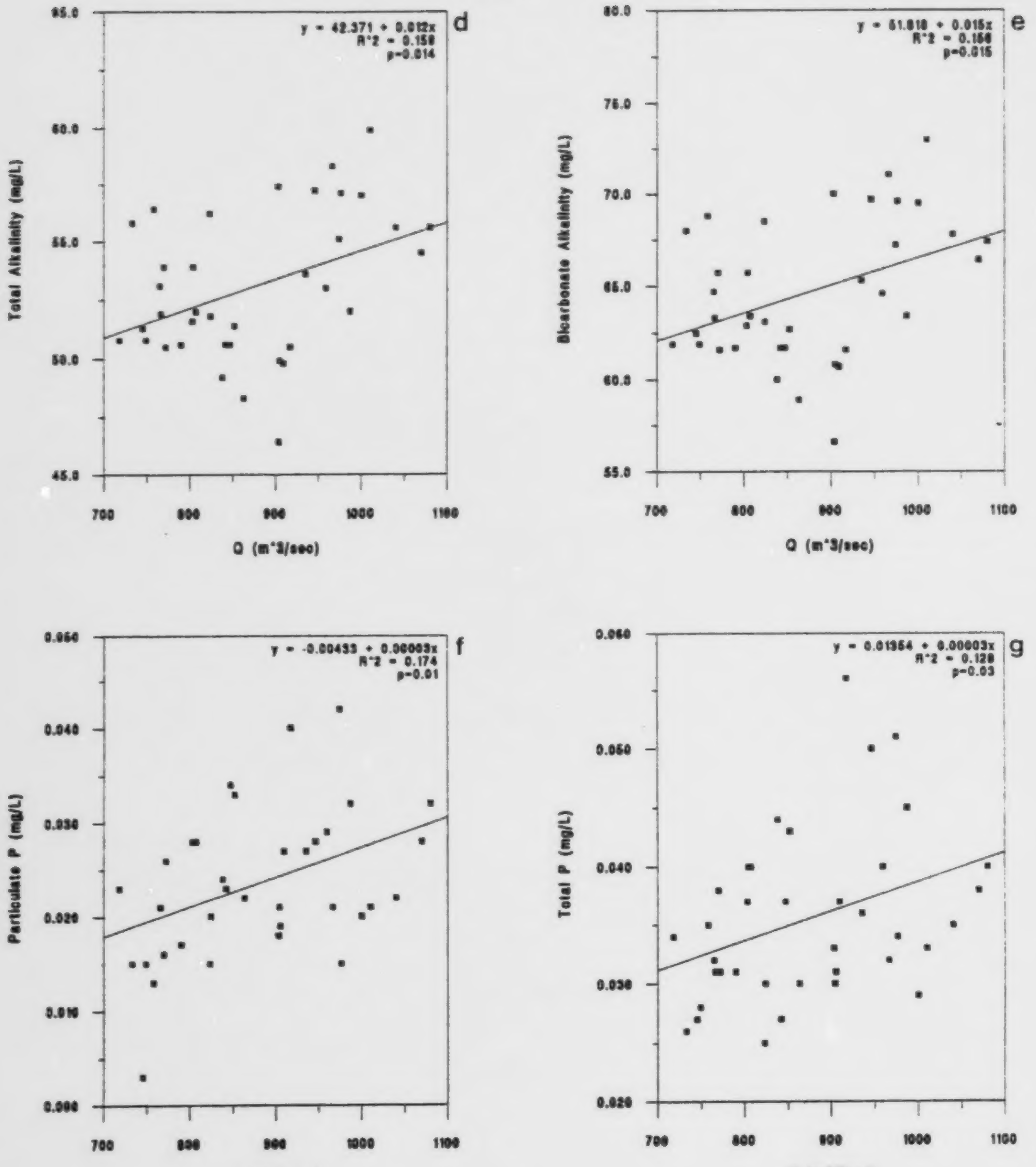


Figure 29. Water chemistry - water discharge relationships in the Burntwood River at Thompson, 1987-1989. a. total NFR b. fixed NFR c. lab turbidity d. total alkalinity e. bicarbonate alkalinity f. particulate P g. total P h. dissolved K i. dissolved Na j. total Cu k. extractable Fe l. extractable Mn m. dissolved Se n. total Zn.



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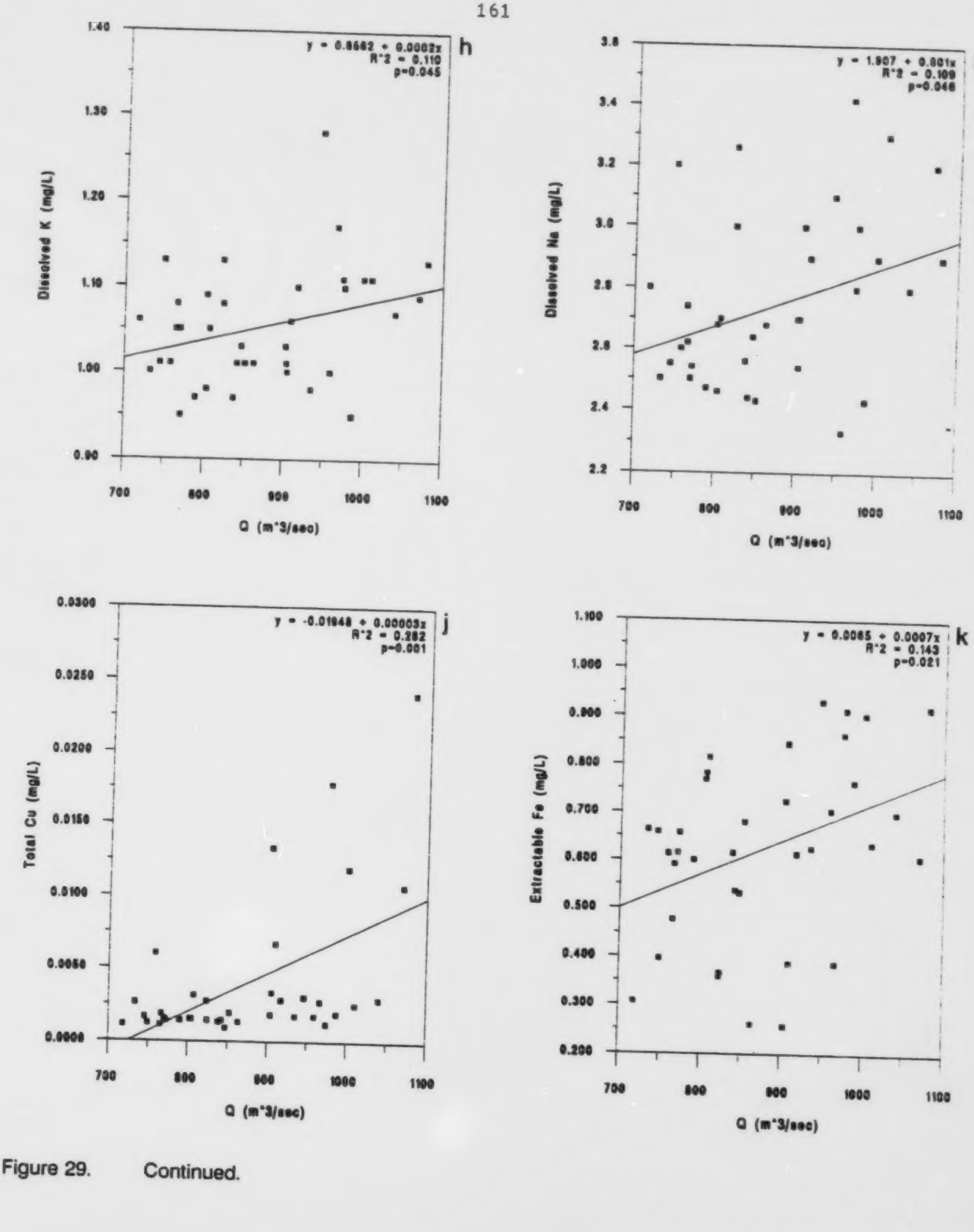

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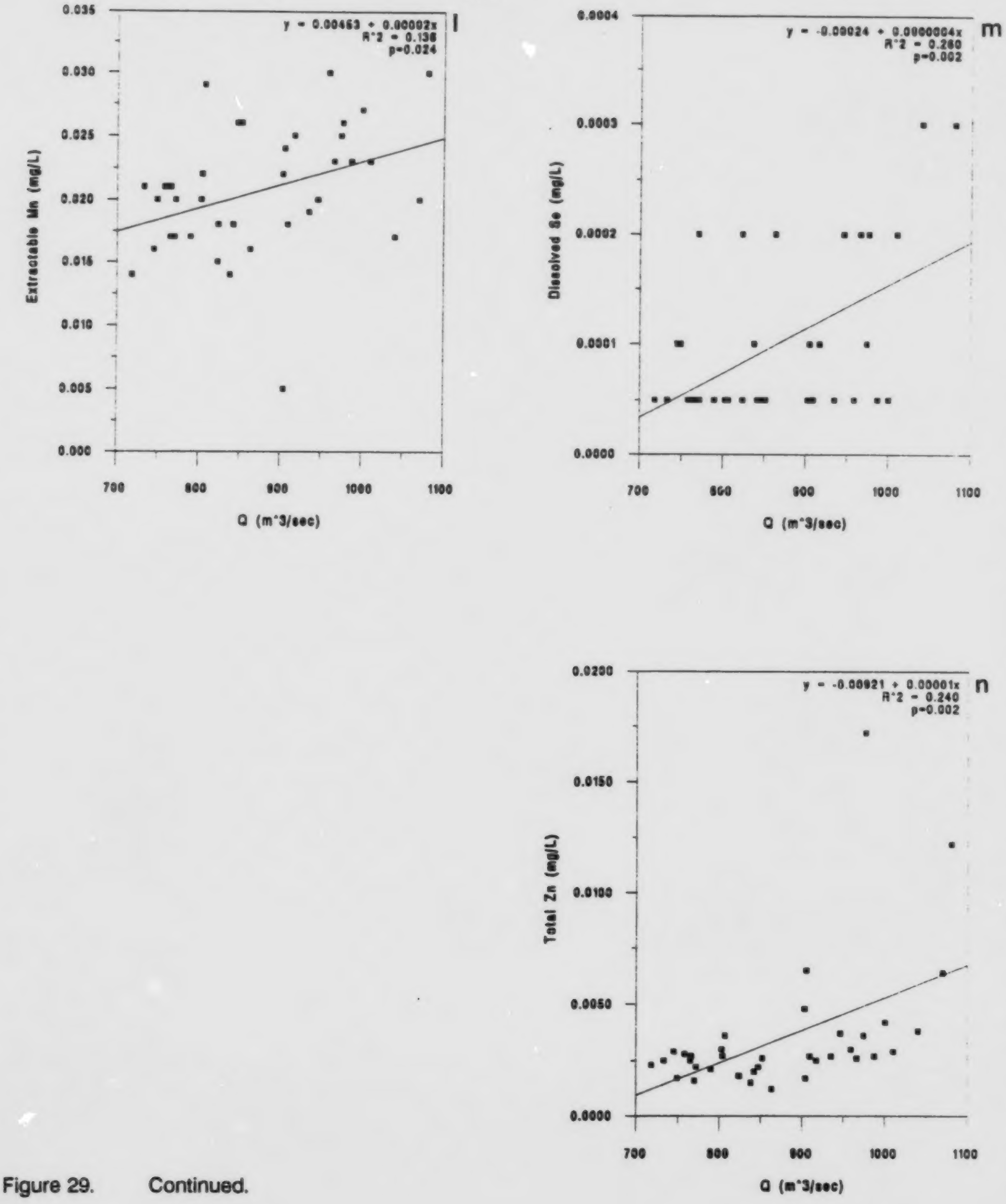

Figure 29. Continued.

total P, dissolved K and Na, total Cu and Zn, extractable Fe and Mn, and dissolved Se.

The shift in the slopes of the alkalinity and Na relationships from negative to positive indicates an important change in the factors regulating concentrations of these parameters. Increasing flows generally had a diluting effect prior to diversion. The current positive correlations indicate an accelerating rate of supply with increasing discharge. Increasing rates of bank and shoreline erosion with increasing discharge are the most probable source of these materials.

The occurrence of positive correlations with discharge for NFR and turbidity in the Burntwood River at Thompson but not upstream at the outlet of Threepoint Lake is indicative of shoreline erosion occurring between these points. Erosion hotspots have been identified on the southern shore of Wuskwatim Lake and along the Burntwood River between Gods and Early Morning rapids (Kellerhals 1988).

Few of these significant correlations shows much promise for the prediction of impacts which might result from a change in the operation regime of the diversion or for management of impacts related to the current operation regime. Although more variables were correlated with discharge during the FEMP study, the relationships generally were much weaker than in the preceding periods. Regression on discharge accounted for 20% or less of the variance in alkalinity (total and bicarbonate), P (particulate and total), dissolved K and Na, and extractable Fe and Mn. The general occurrence of weaker correlations following diversion is probably a result of the smaller range of fluctuation in discharge, relative to the mean discharge, than during the pre-development period. The coefficient of variation (c.v.) of daily discharge was 122% of the mean for the period between 1967 and 1978, compared with a c.v. of just 12% of the mean during the 1987-1989 FEMP study period.

It is unlikely that any of the discharge relationships for the FEMP study period will be of use for predictive purposes due to the substantial portion of total variance (80-90%) that remains unaccounted for. Although 21-30% of the variance in total Cu and Zn and dissolved Se was accounted for by regression on discharge, the utility of these relationships is questionable because the correlations were driven by a few points rather than representing a general linear trend in the data. Only the discharge relationships for NFR (total and fixed) and turbidity appear to have predictive value as they account for 22-42% of the variance in the dependent variables and the linear model represents the general trend in the data.

3.4.3 Seasonal Differences

As was the case at many other sites, chlorophyll a concentrations were generally lower under ice-cover than during the open water season (Figure 30). This probably reflects the reduced light penetration through the snow and ice. There were no other differences in water quality between the open water and ice-cover seasons during the 1987-1989 FEMP study. No other variables

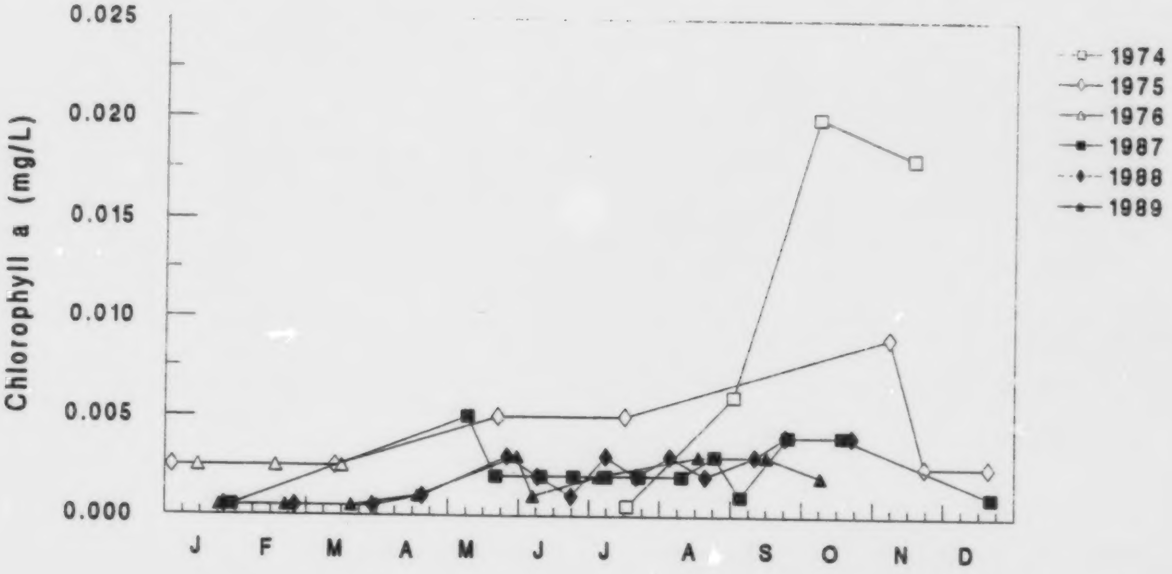

Figure 30. Seasonal trends in chlorophyll a concentrations in the Burntwood River at Thompson.

followed any definable seasonal pattern. This trend of similar water quality in summer and winter was prevalent during the pre-development and construction periods as well.

3.4.4 Changes in Water Quality Since Churchill River Diversion

Significant changes associated with the construction period included a 64% increase in turbidity and a 12% increase in hardness (Table 20). Turbidity remained at or near this elevated level after the diversion began operating. Conversely, hardness declined to the pre-development level in the immediate post-diversion period, with a further decline occurring after 1978. Hardness was 50% lower in the 1987-1989 period than before diversion.

The only significant change between the pre-development and immediate post-diversion periods was a 230% increase in total P (Table 20). This change coincided with the start of CRD operation, as there was no difference in total P concentration between the pre-development and construction periods. Marginally significant changes between these periods included a 19% increase in TOC and a 29% increase in turbidity.

Considerably more changes in water quality become evident when the FEMP study results are compared with the pre-development data. Turbidity was 94% higher during the FEMP study, while conductivity, alkalinity, and hardness were about 20% lower; dissolved Ca and SO_4 were 32 to 34% lower; TOC, reactive Si, and dissolved Fe were 45 to 49% lower; and dissolved NO_2NO_3 was 84% lower (Table 20). The decrease in conductivity was first evident between the construction

Table 20. Evaluation of the significance of changes in selected water quality parameters in the Burntwood River at Thompson following Churchill River diversion. Values listed are the mean and standard deviation (s.d.) for the respective period and the p value from the Tukey/HSD pairwise comparison. Pre denotes the pre-development period from 1967 to May 1974. Const denotes the construction period between June 1974 and August 1976. Post is the immediate post-diversion period from September 1976 to December 1977. Current represents the FEMP study period between January 1987 and October 1989.

pH (Lab)

	Mean	s.d.	Pre	Const	Post
Pre	7.66	0.15			
Const	7.67	0.06	1.000		
Post	7.45[1]	0.07	0.310	0.109	
Current	7.70	0.10	0.981	0.979	0.120

Conductivity (Lab)

	Mean	s.d.	Pre	Const	Post
Pre	141.0[1]	7			
Const	148.0[1]	6	0.822		
Post	125.0	25	0.137	0.047	
Current	115.0	2	0.031	0.013	0.459

Turbidity (Lab)

	Mean	s.d.	Pre	Const	Post
Pre	14.0[1]	2			
Const	23.0	1	0.001		
Post	18.0	<1	0.075	0.083	
Current	27.0[1]	2	<0.001	0.249	0.001

Alkalinity (Total)

	Mean	s.d.	Pre	Const	Post
Pre	65.0[1]	3			
Const	70.0[1]	4	0.205		
Post	60.0[1]	2	0.132	0.004	
Current	53.0[1]	1	0.003	<0.001	0.034

Hardness (Total)

	Mean	s.d.	Pre	Const	Post
Pre	69.0[1]	3			
Const	77.0[1]	4	0.043		
Post	66.0[1]	<1	0.661	0.004	
Current	53.0	<1	<0.001	<0.001	<0.001

NFR (Lab)

	Mean	s.d.	Pre	Const	Post
Pre	11.0	10			
Const	8.0	6	0.939		
Post	13.0	4	0.999	0.873	
Current	16.0[1]	1	0.890	0.438	0.946

TOC

	Mean	s.d.	Pre	Const	Post
Pre	12.63	1.11			
Const	14.00	1.00	0.335		
Post	15.00	1.41	0.074	0.950	
Current	6.94	0.33	0.001	<0.001	<0.001

Diss NO_2NO_3

	Mean	s.d.	Pre	Const	Post
Pre	0.140	0.159			
Const	i.d.				
Post	i.d.				
Current	0.022	0.006	0.029		

Total P

	Mean	s.d.	Pre	Const	Post
Pre	0.028	0.009			
Const	0.039	0.010	0.351		
Post	0.065	0.007	0.002	0.016	
Current	0.036[1]	0.003	0.665	0.970	0.016

Table 20. continued.

Reactive Si	Mean	s.d.	Pre	Const	Post
Pre	3.94^1	0.25			
Const	4.10	0.76	0.973		
Post	3.25^1	0.50	0.117	0.062	
Current	2.01	0.12	0.001	0.001	0.005

Diss Na	Mean	s.d.	Pre	Const	Post
Pre	2.5^1	0.2			
Const	2.8^1	0.2	0.095		
Post	2.5	<0.1	1.000	0.081	
Current	2.7^1	0.1	0.349	0.948	0.313

Diss K	Mean	s.d.	Pre	Const	Post
Pre	1.00	0.12			
Const	1.06	0.10	0.928		
Post	1.25	0.35	0.111	0.408	
Current	1.06^1	0.02	0.959	1.000	0.484

Diss Mg	Mean	s.d.	Pre	Const	Post
Pre	i.d.				
Const	6.5^1	0.1			
Post	i.d.				
Current	4.0	0.1		<0.001	

Diss Ca	Mean	s.d.	Pre	Const	Post
Pre	21.3	2.7			
Const	21.5	0.8	0.996		
Post	i.d.				
Current	14.6	0.1	0.009	0.011	

Diss SO_4	Mean	s.d.	Pre	Const	Post
Pre	4.1	0.2			
Const	4.0	0.3	0.995		
Post	4.0	0.6	0.996	1.000	
Current	2.7	0.3	0.007	0.009	0.017

Diss Cl	Mean	s.d.	Pre	Const	Post
Pre	1.2	0.2			
Const	2.7	2.2	0.279		
Post	i.d.				
Current	1.2	0.1	0.983	0.177	

Diss F	Mean	s.d.	Pre	Const	Post
Pre	i.d.				
Const	0.06	i.d.			
Post	0.12	0.07			
Current	0.09	0.01			0.291

Extr Fe	Mean	s.d.	Pre	Const	Post
Pre	i.d.				
Const	0.813	0.050			
Post	0.786	0.008		0.826	
Current	0.628^1	0.078		0.096	0.046

Table 20. continued.

Diss Fe						Extr Mn						Extr Al					
	Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post		Mean	s.d.	Pre	Const	Post
Pre	0.138	0.027				Pre	0.005	i.d.				Pre	i.d.				
Const	0.150	i.d.				Const	0.037	0.005				Const	0.870	0.207			
Post	i.d.					Post	0.030	0.001		0.585		Post	0.545	0.021		0.067	
Current	0.077	0.010	0.022			Current	0.021	0.002		0.013	0.043	Current	0.488	0.095		0.029	0.744

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[1] Discharge-standardized variables.

and immediate post-development periods, declining further between December, 1977, and January, 1987.

Several variables were not routinely measured until the construction period. Among these, significant reductions in dissolved Mg, by 38%, and in extractable Mn and Al, by 44%, occurred between the 1974-1976 construction period and the 1987-1989 FEMP study (Table 20).

The increase in turbidity during the construction period was not simply a result of the reduced flows. Based on the turbidity-discharge relationship for the pre-development period (Figure 25d), the 30% decrease in flow should have produced a 16% decline in turbidity rather than the observed 64% increase. Although not accounted for by the reduced flow volume, the increase in turbidity was indirectly related to the flow reduction which was caused by the damming of the Rat River at the outlet to Notigi Lake. During the pre-development period, Rat River turbidity was about 50% of that in the Burntwood River above Threepoint Lake (Tables 10 and 16) so that removal of this dilution source would be expected to increase turbidity in the Burntwood River below the confluence with the Rat River. However, loss of Rat River flows during the construction period accounts for an increase in turbidity of only 15% (i.e., loss of 30% of total flow with turbidity 50% of that in Burntwood R.; 0.3 x 0.5 = 0.15) leaving most of the increase unaccounted for. Possible causes include some form of construction activity, such as the stabilization of river banks in the vicinity of Thompson, dike construction upstream near Manasan Falls, or the instream work in construction of the Notigi Dam, as well as a natural increase in turbidity in the Burntwood River above Threepoint Lake. None of these causes can be verified due to the lack of data for the Rat River between Notigi and Thompson and the Burntwood River between Threepoint Lake and Thompson during the construction period.

The persistence of elevated turbidity at Thompson beyond the construction period was partly due to the introduction of more turbid water from the Rat River but the primary cause was bank and shoreline erosion below Threepoint Lake. Turbidity in the Rat River above Threepoint Lake was 250% higher during the FEMP study than before diversion, but this was just sufficient to raise turbidity to the natural level in the Burntwood River above Threepoint Lake (Table 10). This increase in Rat River turbidity was probably due to shoreline erosion in the southern region of Notigi Reservoir and to bank and shoreline erosion which may be occurring on the river between the Notigi Dam and Threepoint Lake. Despite the increased turbidity in the Rat River, turbidity at both sites was still 60% lower than at Thompson, indicating elevated turbidity in the Rat River is not the primary cause of the elevated turbidity at Thompson. None of this additional turbidity was generated within Threepoint Lake, as indicated by the similar values in the Rat and Burntwood rivers above Threepoint Lake and in the Burntwood River below Threepoint Lake (Tables 6, 15, and 18). Erosion along the Burntwood River between the lake outlet and Thompson must be responsible. Potential sources include documented erosion sites on the south shore of Wuskwatim Lake and on the Burntwood River between Early Morning Rapids and Gods Rapids (Kellerhals 1988).

The elevation in hardness during construction is directly attributable to the reduction in river flows during this period. The discharge-hardness relationship for the construction period (Figure 27c) predicts a mean hardness of 70 mg L^{-1} at the mean discharge for the pre-development period (1967-1973; 126 m^3 sec^{-1}; Water Survey of Canada 1990). This compares favourably to the actual pre-development mean hardness of 69 mg L^{-1} (Table 20).

The declines in conductivity, alkalinity, hardness, dissolved Ca, TOC, and reactive Si are a direct reflection of the lower values of these variables in the Churchill River at Leaf Rapids (Table 17), indicating diversion was the cause. The apparent decline in Mg between the construction and current study periods also is explained by a lower concentration in the Churchill River. The annual mean Mg concentration at Opachuanau Rapids, on the Churchill River at the inlet to Southern Indian Lake, was 2.72 mg L^{-1} in 1972 (Cleugh 1974), compared with 6.5 mg L^{-1} in the Burntwood River during the construction period (Table 20). The decline in SO_4 was of the same magnitude as the declines in the natural rivers and, therefore, was attributed to a change in methodology. It clearly is not due to diversion. The decrease in TOC was the product of lower TOC concentrations in the Churchill River and of additional losses which occur as the water flows through SIL and Notigi Reservoir (Section 3.3.1.4).

In addition to the present study, the effect of CRD on water quality in the Burntwood River at Thompson has been examined in some detail in two previous studies. Playle and Williamson (1986) evaluated the significance of changes in 16 water quality parameters between the pre-diversion period, represented by data from January 1972 through June 1976, and the post-diversion period which included data from July 1976 to March, 1984. Guilbault et al. (1979) examined changes in four water quality variables (total P, dissolved Ca and K, extractable Al) related to CRD. They considered data from the beginning of the NAQUADAT data record in 1967 (or later, depending on the variable) through 1977. The division point between their pre- and post-diversion periods was not defined in their report.

Playle and Williamson (1986) identified significant changes in 7 variables following Churchill River diversion. As in the present study, they found significant decreases in conductivity, by 21%, and in alkalinity, Ca, and Mg by 15-17%. The magnitudes of the changes in conductivity and alkalinity are about the same as found in this study, while their reported decreases in Ca and Mg were about half those reported here. They also found a significant increase in total P, by 66%, and a small but significant 9% decrease in TIC. In contrast to the present study, Playle and Williamson (1986) reported a significant increase in NFR, by 73%, and no change in turbidity or hardness.

The discrepancies between the studies can be explained by differences in the periods over which changes were examined and the way in which the data records were divided. Playle and Williamson (1986) included the construction period (June 1974 through August 1976) as baseline data and did not consider data prior to 1972. Thus the construction period, in which a significant increase in turbidity occurred, represented about half their pre-development data record, obscuring any change (Figure 31b). This illustrates the importance of stratifying the data as much as possible in the initial analysis to consider the effects of all phases of a development. Data from different phases can always be pooled after it has been determined there are no significant differences.

The finding by Playle and Williamson (1986) of a significant increase in NFR following diversion appears to be a result of the restriction of their database to measurements from 1972 and later. When the entire pre-development data record for 1968 through May 1974 is included in the analysis, the difference disappears. NFR concentrations generally tended to be lower and were much less variable from 1972 to 1976 than between 1968 and 1972 (Figure 31a). There does not appear to be any justification for the exclusion of the pre-1972 data on the basis of sampling or analytical incompatibility. It must be concluded there has been no detectable change in annual mean NFR related to CRD.

Notwithstanding the lack of a significant change in mean NFR, it appears from Figure 31a that the minimum NFR has increased since diversion. This is most apparent in the measurements between 1987 and 1989. During this period, NFR never was lower than 7 mg L^{-1} while 19 of the 45 (42%) measurements made before September, 1976,

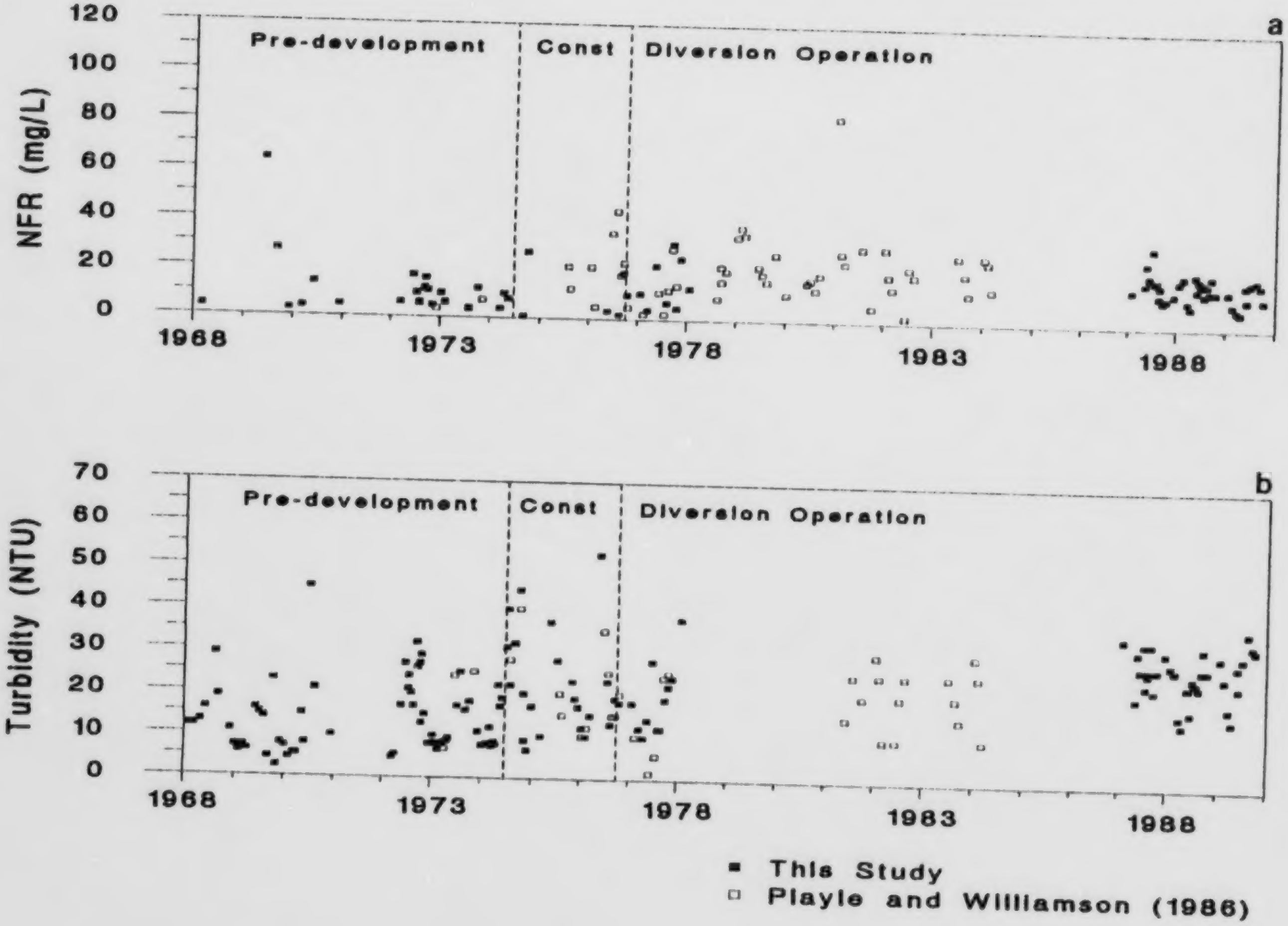

Figure 31. Temporal trends in (a) NFR, (b) turbidity, (c) alkalinity, (d) hardness, and (e) total P in the Burntwood River at Thompson based on data analyzed by Playle and Williamson (1986) and data abalyzed in this study.

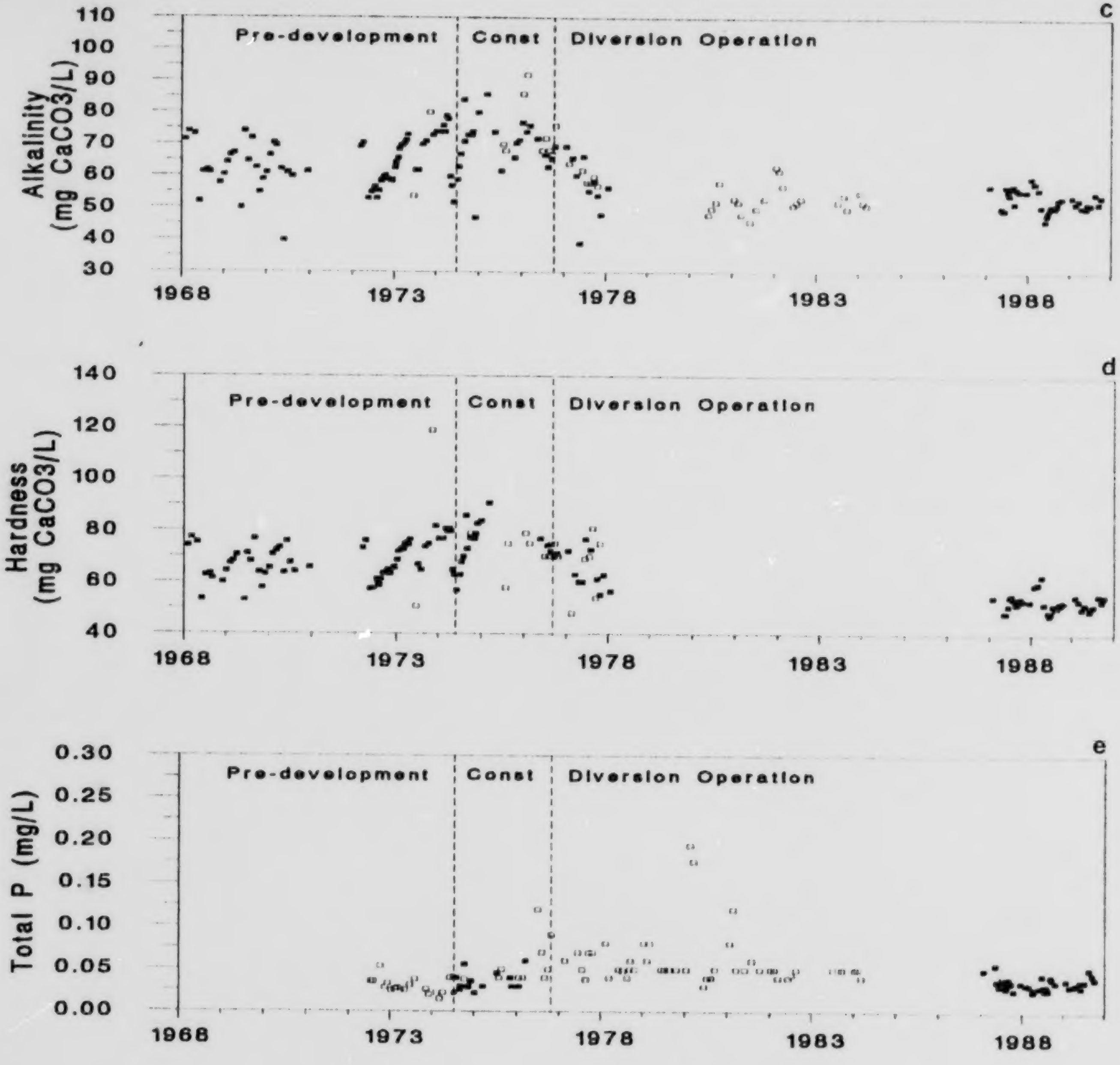

Figure 31. Continued.

were between 1 and 7 mg L^{-1}.

Playle and Williamson (1986) identified the paradox of elevated NFR in the absence of a change in turbidity. Although it now appears that it is turbidity which increased significantly in the absence of a change in NFR, the paradox remains unresolved. One explanation for this apparent contradiction is the extreme variability in NFR compared to turbidity, particularly in the pre-development period (Figure 31 a and b). The detection of a significant difference therefore requires a smaller change in turbidity than in NFR. Both parameters may show the same general trend over time, as appears to be the case, but only one will show a significant increase. Turbidity may provide a better indication of the change in suspended sediment levels at this site due to CRD because of the more extensive pre-development data record for this variable. Over 100 measurements of turbidity were taken prior to the start of construction compared with 45 NFR measurements, therefore mean turbidity is less strongly influenced by occasional high values.

The lack of a significant change in hardness in the analysis by Playle and Williamson (1986) clearly was a result of the short data record available to them at the time of their study (Figure 31d). When the FEMP study data are added to this record, the decrease becomes apparent.

The increase in total P reported by Playle and Williamson (1986) appears to be a valid conclusion based on the analysis presented here (Table 20) and the data in Figure 31e. Similar increases were observed in the back-flooded area of Stephens Lake (Hecky and Harper 1974) and at Southern Indian Lake and other sites on the CRD immediately after impoundment (R.E. Hecky, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.). The elevated total P levels probably were a result of the release of P from flooded soils and vegetation through leaching and decomposition. The flux of P from the flooded materials upstream may appear to have been relatively short-lived, peaking in about 1980 and returning to pre-development values some time between 1981 and 1987. The results of this study indicate total P concentrations in the 1987-1989 study period were not significantly different from the levels prior to diversion. However, the P flux may still be occurring, as indicated by higher total P concentrations in the Rat and Burntwood rivers than in the Churchill River. The end point of the process is not the pre-development concentration for the site, but is the level in the source of flow. A long-term impact of CRD will be a net decrease in the total P concentration in the Rat and Burntwood rivers. Based on the rate of decline in total P from the peak of 0.083 mg L^{-1} in 1980 to 0.036 mg L^{-1} during the 1987-1989 FEMP study, the annual mean concentration in the Burntwood River should reach that in the Churchill River (0.017 mg L^{-1}) within the next decade. This can only be confirmed by repeated monitoring at intervals of about 5 years.

Guilbault et al. (1979) reported a decrease in dissolved Ca, as found in this study and by Playle and Williamson (1986), as well as increases in dissolved K and extractable Al which were not detected in the other studies. In the present study, a significant decrease in extractable Al was found between the construction and current periods using the same pre-diversion data set analyzed by Guilbault et al. (1979). This discrepancy appears to be an artefact of the previous study having compared dissolved Al in the pre-diversion period with post-diversion measurements of extractable Al. Even when measured in the same sample, the extractable Al concentration will be much higher than the dissolved Al concentration because the extractable value includes Al bound to particulates while the dissolved value only includes Al in solution. In the current study, the extractable Al concentration in the Burntwood River at Thompson was, on average, about 5 times higher than the dissolved Al concentration (Table 19). This confusion of parameters is indicated by the number of pre-development extractable Al measurements, 5, and the mean, 0.04 mg L^{-1}, stated by Guilbault et al. (1979). Our review of the database found 5 measurements of dissolved Al during the construction period, with a mean of 0.04 mg L^{-1}, compared with 12 extractable Al values with a mean of 0.870 mg L^{-1}. No extractable or dissolved Al measurements were made in the pre-development period. Not only is there no evidence to support and increase in extractable Al, there is no mechanism to explain such an increase. Between 1987 and 1989, extractable Al concentrations in the Rat and Burntwood rivers above Threepoint Lake were similar (Tables 6 and 15). A significantly higher concentration in the Rat River would be required to support the occurrence of a significant increase due to diversion.

Although a significant decline in extractable Al occurred between the construction and current study periods, it cannot be concluded that concentrations have declined from the natural level, in the absence of pre-development data for Al or any correlate. The upstream data collected in the FEMP study suggest a more reasonable conclusion might be that extractable Al levels increased during the construction period. Concentrations in the Burntwood and Rat rivers above Threepoint Lake between 1987 and 1989 were not only similar to each other, but were about the same as measured at Thompson as well (Tables 6, 15, and 19). The only way a lower concentration could have developed at Thompson is if the level in the Burntwood River above Threepoint Lake was higher during the pre-development period. Given the general stability of water chemistry in the upper Burntwood River (Section 3.1.4.1), this seems unlikely. Thus, it is the higher concentration during construction, rather than the lower current value, which appears to require explanation.

The finding of a significant increase in dissolved K following diversion by Guilbault et al. (1979) was most probably an artefact of their approach to the statistical analysis. The individual measurements over time were treated as replicates which may have violated the assumption of independence of observations required for valid application of Student's t-test (Snedecor and Cochran 1980). Time-series measurements are typically inter-dependent due to seasonal variations in environmental factors (Lettenmaier 1976). The use of dependent observations as replicates has the same effect as using the same measurement more than once, artificially increasing the sample size and decreasing the variance. This dependence of observations can be avoided by calculating means for relatively long periods. Annual means of time-series measurements, as used in this study, generally are not inter-correlated and, therefore, can be used as replicates in parametric statistical analyses (Lettenmaier 1976). When the analysis is repeated on the same data set using annual means as replicates, no significant change is found (Table 20).

3.4.5 Effect of CRD on Heavy Metals

The erosion of inorganic sediments and associated heavy metals introduces the potential for the occurrence of elevated concentrations of metals in the water. This concern does not apply to all metals however, but only to those for which the eroding shoreline materials represent an important source. In the Burntwood River at Thompson, the eroding sediments appear to be a source of Fe, Mn, and Zn, as indicated by their positive correlations with river discharge. Despite these contributions through erosion, concentrations of Fe and Mn did not become elevated following CRD. Instead, concentrations of these metals were lower during the FEMP study than during the construction period (Table 20). This paradox is explained by 32-54% lower Fe and Mn concentrations in the Rat River (Table 15), which now dominates flows on the lower Burntwood River, than in the upper Burntwood River which dominated flows prior to diversion (Table 10). Concentrations of most metals are lower in the Rat River than in the Burntwood River (Tables 10 and

15), thus the CRD was largely self-mitigating with respect to the impact of erosion on the concentrations of heavy metals.

Although concentrations of metals have not increased due to CRD, it should be kept in mind that the mass flux of metals down the Burntwood River has increased substantially due to the large increase in river flow. It is not known if the increased loading will have an impact on metal availability at sites downstream, and particularly at sites of sediment deposition.

3.4.6 Adverse Effects

The only adverse effect of Churchill River diversion on physical and chemical water quality in the Burntwood River near Thompson has been the 100% increase in turbidity caused primarily by the erosion of glacial clays along the Burntwood River between Threepoint Lake and Thompson. The increase in turbidity exceeded the increase permitted under the Canadian Water Quality Guidelines for recreational water quality and aesthetics by more than a factor of 2. These guidelines allow for a 5.0 NTU increase in turbidity in waters such as the Burntwood River where turbidity is <50 NTU. Thus, the primary impact of CRD has been visual and physical, causing the water to appear "muddier" than it did prior to diversion and probably increasing the cost of turbidity reduction for those communities which use the Burntwood River for drinking water.

Although the safety of the water for human consumption has not been compromised by CRD, such an obvious visual change may have caused water users to become concerned about the safety of their drinking water source. Over 10 years passed between the first increase in turbidity during construction of the diversion and the first detailed analysis of changes by Playle and Williamson (1986). This has left much time for local residents to wonder about, and develop their own opinions regarding, the nature and significance of changes which have occurred in their water source. Indeed, turbid water has been equated with poor quality water by some northern residents, while clear water is thought to be safe for drinking (Seidl et al. 1990). This study has shown the increased turbidity caused by CRD has not affected the quality of water in the Burntwood River for human consumption other than the increase in turbidity itself.

It could be argued that all of the other significant changes in water quality since CRD have been beneficial, actually improving the quality of water for domestic uses. The water is now slightly softer and has a lower major ion content although these improvements are small compensation for the degraded appearance of the river and the increased cost of drinking water treatment.

Algal standing crops and rates of photosynthesis in the Burntwood River below Thompson were severely light-limited prior to diversion (Hecky and Harper 1974). The increased turbidity due to diversion therefore may not have had a significant effect on the primary producers in this reach of the Burntwood River.

3.5 Split Lake Area

3.5.1 Burntwood River at Inlet to Split Lake

3.5.1.1 Current Water Quality

Water quality at this site was typical for the Burntwood River below the confluence with the CRD. Concentrations of all but two parameters were the same as at the Thompson site (Table 21). These exceptions were the annual mean total and fixed NFR concentrations which were 45-50% higher at the inlet to Split Lake than at Thompson. This may be a product of the documented bank erosion site near First Rapids (Kellerhals 1988).

The same variables exceeded the guideline values for drinking water, recreation, and aquatic life at this site as found in the natural rivers (Table 22). All measurements of turbidity always exceeded the drinking water guideline, and the maximum values also exceeded the guideline for recreational use. Mean colour exceeded the drinking water guideline by about 47%. Maximum extractable Mn exceeded the drinking water guideline by a factor of 3.5 and the mean was at the guideline level. Extractable Fe exceeded the guidelines for both drinking water and aquatic life by a factor of 2.6 on average and by as much as a factor of 9.5. Extractable Al exceeded the aquatic life guideline by a factor of 5.9 on average and by as much as a factor of 21. The total Cu concentration was 2.4 times the aquatic life guideline on average with maximum concentrations as much as a factor of 14 above the guideline value. Peak total Cd and total Pb also exceeded the aquatic life guidelines.

Table 21. Comparison of current water quality in the Burntwood River at Thompson and at the inlet to Split Lake. X_{thm} denotes the 1987-1989 mean for the Burntwood River at Thompson. X_{spl} denotes the 1987-1989 mean for the Burntwood River at the inlet to Split Lake. p is the probability of Student's t-statistic for the comparison.

Location		pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Colour Relative	TDS $mg\ L^{-1}$	NFR $mg\ L^{-1}$	NFR Fixed $mg\ L^{-1}$	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	Diss Ca $mg\ L^{-1}$
Burntwood R.	p	0.954	0.761	0.201	0.988	0.704	0.015	0.026	0.727	0.525	0.156
	X_{thm}	7.69	115.	27.0[1]	21.	60.0	16.3[1]	14.7[1]	53.0[1]	53.0	14.6
	X_{spl}	7.76	123.	35.0[1]	22.	64.7	24.3	21.3	56.7[1]	56.7	15.9

	Diss Mg $mg\ L^{-1}$	Diss K $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Cl $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO_4 $mg\ L^{-1}$	Ortho P $mg\ L^{-1}$	TDP $mg\ L^{-1}$	Tot P $mg\ L^{-1}$
p	0.991	1.000	0.997	0.999	0.998	0.982	0.943	1.000	1.000	0.810
X_{thm}	4.0	1.06[1]	2.7[1]	2.00	1.2	0.09	2.7	0.003	0.012	0.036[1]
X_{spl}	4.1	1.05	2.8	1.96	1.2	0.10	2.9	0.003	0.012	0.039

				TDN $mg\ L^{-1}$	Diss NO_2-NO_3 $mg\ L^{-1}$	Susp N $mg\ L^{-1}$	DOC $mg\ L^{-1}$	TOC $mg\ L^{-1}$	DIC $mg\ L^{-1}$	Chlorophyll a $mg\ L^{-1}$
p				1.000	0.992	0.459	0.716	0.381	0.260	0.990
X_{thm}				0.248	0.022	0.049	6.48	6.94	12.4	0.002
X_{spl}				0.248	0.019	0.063	6.78[1]	7.41[1]	13.4	0.002

Table 21. continued.

Location		Extr Al mg L^{-1}	Diss Al mg L^{-1}	Diss As mg L^{-1}	Tot Cd mg L^{-1}	Diss Cd mg L^{-1}	Tot Co mg L^{-1}	Tot Cu mg L^{-1}	Diss Cu mg L^{-1}	Extr Fe mg L^{-1}	Diss Fe mg L^{-1}
Burntwood R.	p	0.274	0.998	1.000	1.000	1.000	0.835	1.000	0.997	0.393	0.573
	X_{thm}	0.488	0.080	0.0002	0.0001	0.0001	0.0004	0.0042[1]	0.0014	0.628[1]	0.077
	X_{spl}	0.596	0.082	0.0002	0.0001	0.0001	0.0005	0.0047	0.0015	0.787	0.091

	Tot Pb mg L^{-1}	Diss Pb mg L^{-1}	Extr Mn mg L^{-1}	Diss Mn mg L^{-1}	Tot Ni mg L^{-1}	Diss Ni mg L^{-1}	Diss Se mg L^{-1}	Tot V mg L^{-1}	Diss V mg L^{-1}	Tot Zn mg L^{-1}	Diss Zn mg L^{-1}
p	0.994	0.991	0.107	1.000	0.280	0.183	1.000	0.502	1.000	0.890	0.959
X_{thm}	0.0010	0.0010	0.021[1]	0.004	0.0018	0.0006	0.0001[1]	0.0013	0.0004	0.0035[1]	0.0010
X_{spl}	0.0011	0.0007	0.031	0.004[1]	0.0023	0.0011	0.0001[1]	0.0016	0.0004	0.0046	0.0007[1]

[1] Discharge-standardized variables.

Table 22. Summary statistics for physical and chemical water quality in the Burntwood River at the inlet to Split Lake, September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.6	0.2	8.2	7.2
	Lab		7.76	0.19	8.17	7.33
Conductivity	Field	µS cm^{-1}	120	12	136	86
	Lab		123	6	138	107
Turbidity	Field	NTU	34^{+}	11	$70^{+,++}$	17^{+}
	Lab		35^{+}	10	$71^{+,++}$	20^{+}
Colour	True	Relative	22^{+}	6	40^{+}	10
TDS		mg L^{-1}	65	3	71	58
NFR	Total	mg L^{-1}	24	16	107	10
	Fixed	mg L^{-1}	21	15	100	8
Alkalinity	Total	mg L^{-1}	56.9	3.2	64.7	49.1
	Bicarbonate	mg L^{-1}	69.4	3.9	78.9	59.9
Hardness	Total	mg L^{-1}	56.6	3.1	62.9	48.1
	Non-carbonate	mg L^{-1}	0.7	1.0	3.7	0
Free CO_2		mg L^{-1}	2.45	1.79	8.87	0
Phosphorus	Particulate	mg L^{-1}	0.027	0.014	0.088	0.011
	Ortho PO_4	mg L^{-1}	0.003	0.003	0.017	<0.003
	Total Dissolved	mg L^{-1}	0.012	0.006	0.029	<0.003
	Total	mg L^{-1}	0.039	0.013	0.102	0.021
Nitrogen	Particulate	mg L^{-1}	0.062	0.026	0.123	<0.010
	Total Dissolved	mg L^{-1}	0.248	0.039	0.370	0.200
	NO_2-NO_3	mg L^{-1}	0.019	0.025	0.077	<0.010
	Total NH_3	mg L^{-1}	0.014	0.007	0.030	<0.005
	Total	mg L^{-1}	0.31	0.04	0.40	0.24
Carbon	DOC	mg L^{-1}	6.78	0.76	9.10	5.70
	POC	mg L^{-1}	0.635	0.316	2.100	0.270
	TOC	mg L^{-1}	7.40	0.89	9.72	6.11
	DIC	mg L^{-1}	13.4	0.7	14.9	11.8
Chlorophyll-a		mg L^{-1}	0.002	0.001	0.005	<0.001
Ca	Dissolved	mg L^{-1}	15.9	0.9	17.6	13.0
Mg	Dissolved	mg L^{-1}	4.1	0.2	4.6	3.6
K	Dissolved	mg L^{-1}	1.05	0.06	1.21	0.96
Na	Dissolved	mg L^{-1}	2.6	0.3	3.5	2.4

Table 22. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	1.96	0.68	3.39	0.89
Cl	Dissolved	mg L^{-1}	1.2	0.2	1.5	0.9
F	Dissolved	mg L^{-1}	0.09	0.02	0.15	0.08
SO_4	Dissolved	mg L^{-1}	2.9	0.9	7.1	1.5
Al	Extractable	mg L^{-1}	0.592^{+++}	0.315	2.140^{+++}	0.235^{+++}
	Dissolved	mg L^{-1}	<0.100	0.058	0.255	<0.100
As	Dissolved	mg L^{-1}	0.0002	0.0001	0.0003	<0.0001
Ba	Total	mg L^{-1}	<0.08	0.01	0.11	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0003^{+++}	<0.0001
	Dissolved	mg L^{-1}	<0.0001	0.0001	0.0003	<0.0001
Co	Total	mg L^{-1}	0.0005	0.0004	0.0021	<0.0005
Cu	Total	mg L^{-1}	0.0048^{+++}	0.0062	0.0278^{+++}	0.0010
	Dissolved	mg L^{-1}	0.0014	0.0006	0.0032	0.0010
Fe	Extractable	mg L^{-1}	0.774$^{+,+++}$	0.416	2.860$^{+,+++}$	0.299
	Dissolved	mg L^{-1}	0.090	0.053	0.208	<0.007
Pb	Total	mg L^{-1}	0.0010	0.0007	0.0034^{+++}	<0.0007
	Dissolved	mg L^{-1}	0.0007	0.0008	0.0035	<0.0007
Mn	Extractable	mg L^{-1}	0.030	0.027	0.177^{+}	0.004
	Dissolved	mg L^{-1}	0.004	0.002	0.011	<0.002
Ni	Total	mg L^{-1}	0.0023	0.0011	0.0077	0.0010
	Dissolved	mg L^{-1}	0.0011	0.0005	0.0021	<0.0005
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0004	<0.0001
V	Total	mg L^{-1}	0.0016	0.0008	0.0047	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0002	0.0009	<0.0005
Zn	Total	mg L^{-1}	0.0045	0.0033	0.0181	0.0011
	Dissolved	mg L^{-1}	0.0009	0.0007	0.0021	<0.0003
Discharge	Daily	m^3 sec^{-1}	908	105	1131	719

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recreational use.

$^{+++}$ Exceeds limit for protection of aquatic life.

No analysis of changes in water quality since CRD is possible for this site due to the lack of pre-development data.

3.5.1.2 Water Quality - Water Discharge Relationships

Positive correlations with discharge dominated at this site during the 1987-1989 study period, consistent with our findings at sites upstream on the Burntwood River. The positive correlates included turbidity (lab and field), alkalinity (total and bicarbonate), free CO_2, DOC, TOC, and dissolved Na and Se (Figure 32). Negative correlates with discharge were dissolved Mn and Zn. Again, none of the relationships showed much impact prediction or management potential. Nine of the 11 significant correlations accounted for just 10-20% of the variance in the dependent variable, although 2 correlations accounted for 25-35% of the variance.

The most useful of these water quality - water discharge relationships for prediction and management purposes are those involving turbidity and alkalinity (Figure 32). Although none of these correlations accounted for more than 20% of the variance, all reflected a general linear trend in the data. The strength of these relationships might be improved with additional data collection which may reduce the influence of the few outlying values. The discharge relationship for dissolved zinc also has potential for predictive purposes but, with the relatively few points on the curve (Figure 32k; n=14), additional data collection is advised to determine if the relationship holds.

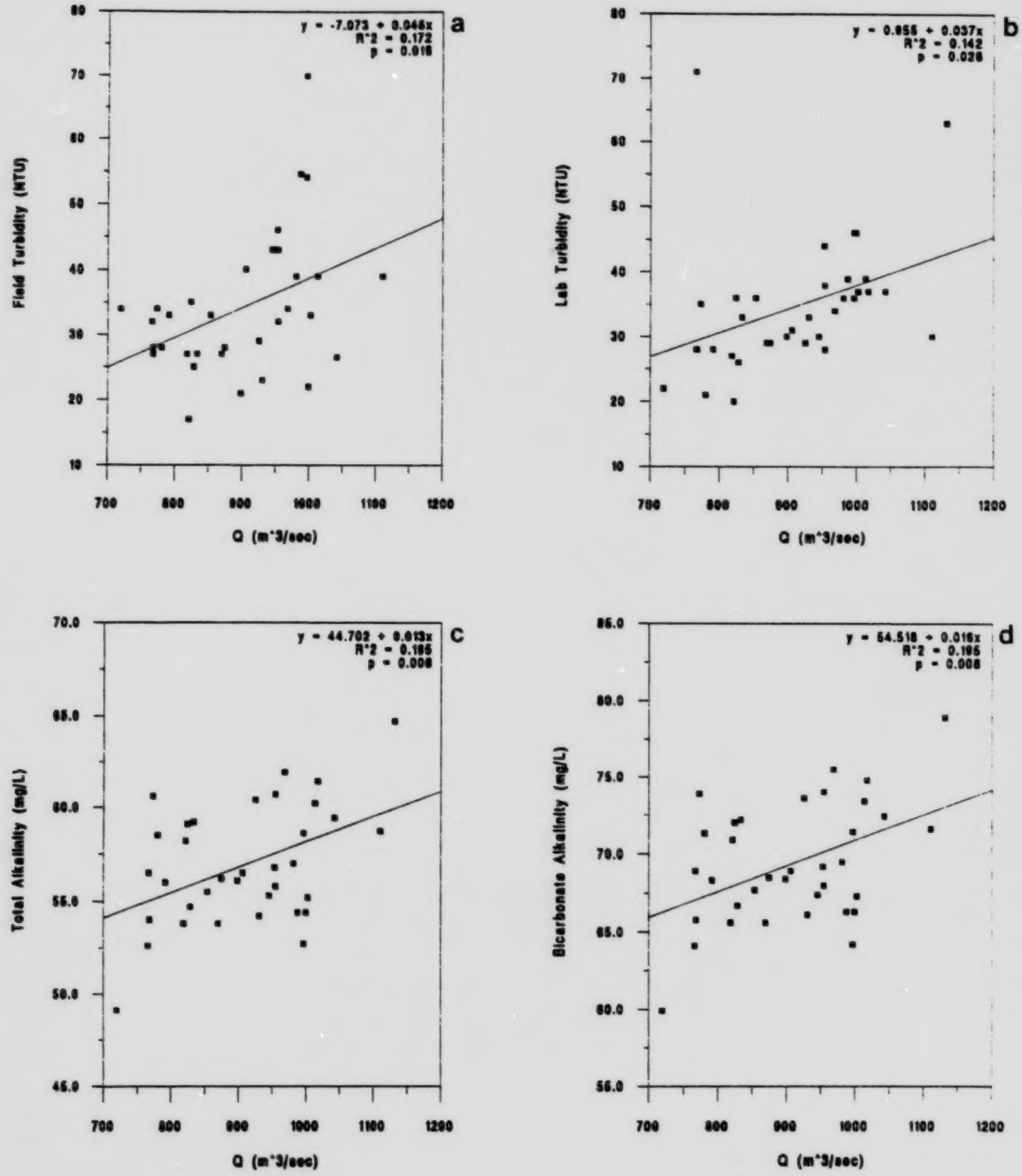

Figure 32. Water chemistry - water discharge relationships in the Burntwood River at the inlet to Split Lake, 1987-1989. a. field turbidity b. lab turbidity c. total alkalinity d. bicarbonate alkalinity e. free CO_2 f. DOC g. TOC h. dissolved Na i. dissolved Mn j. dissolved Se k. dissolved Zn.

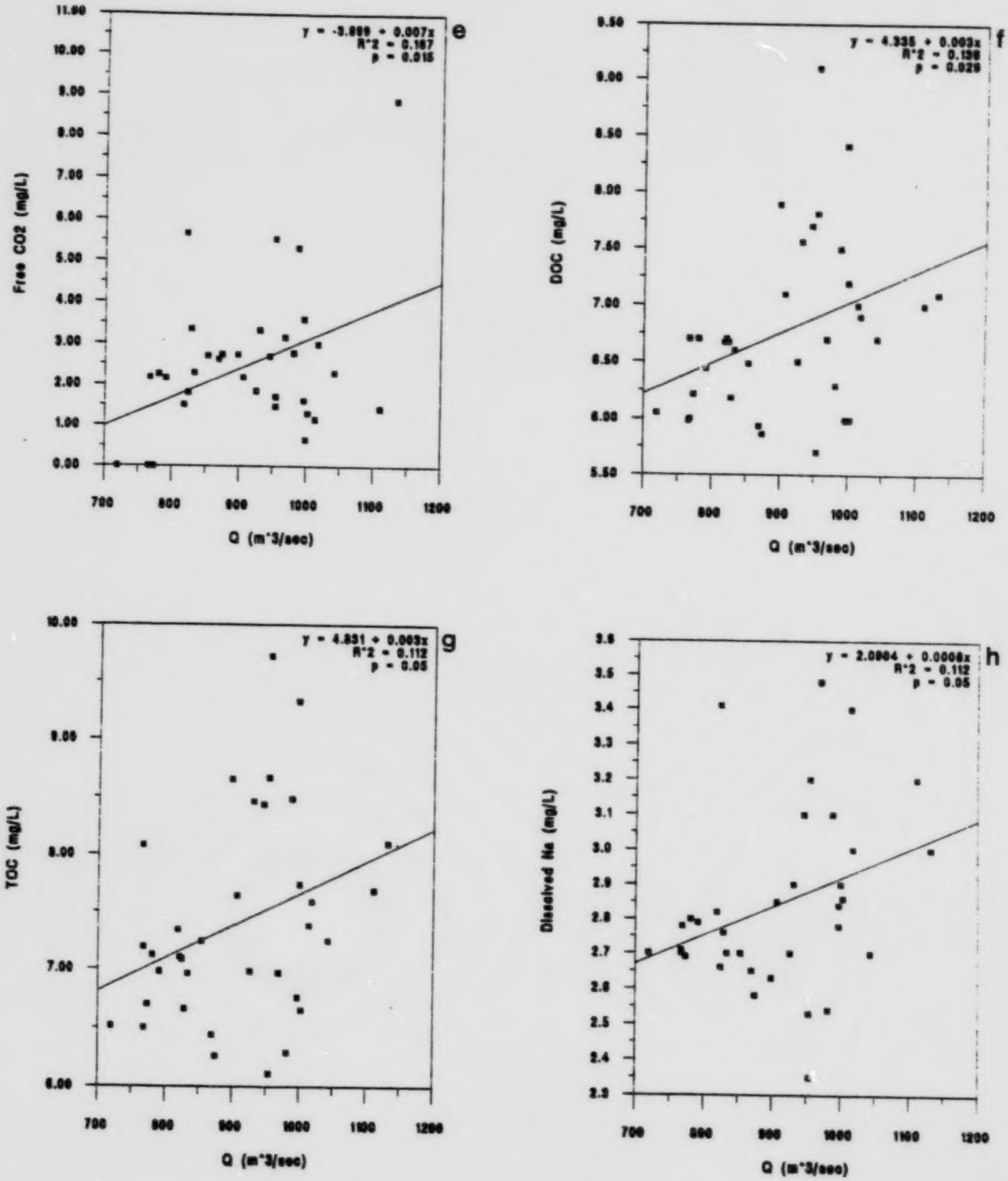

Figure 32. Continued.

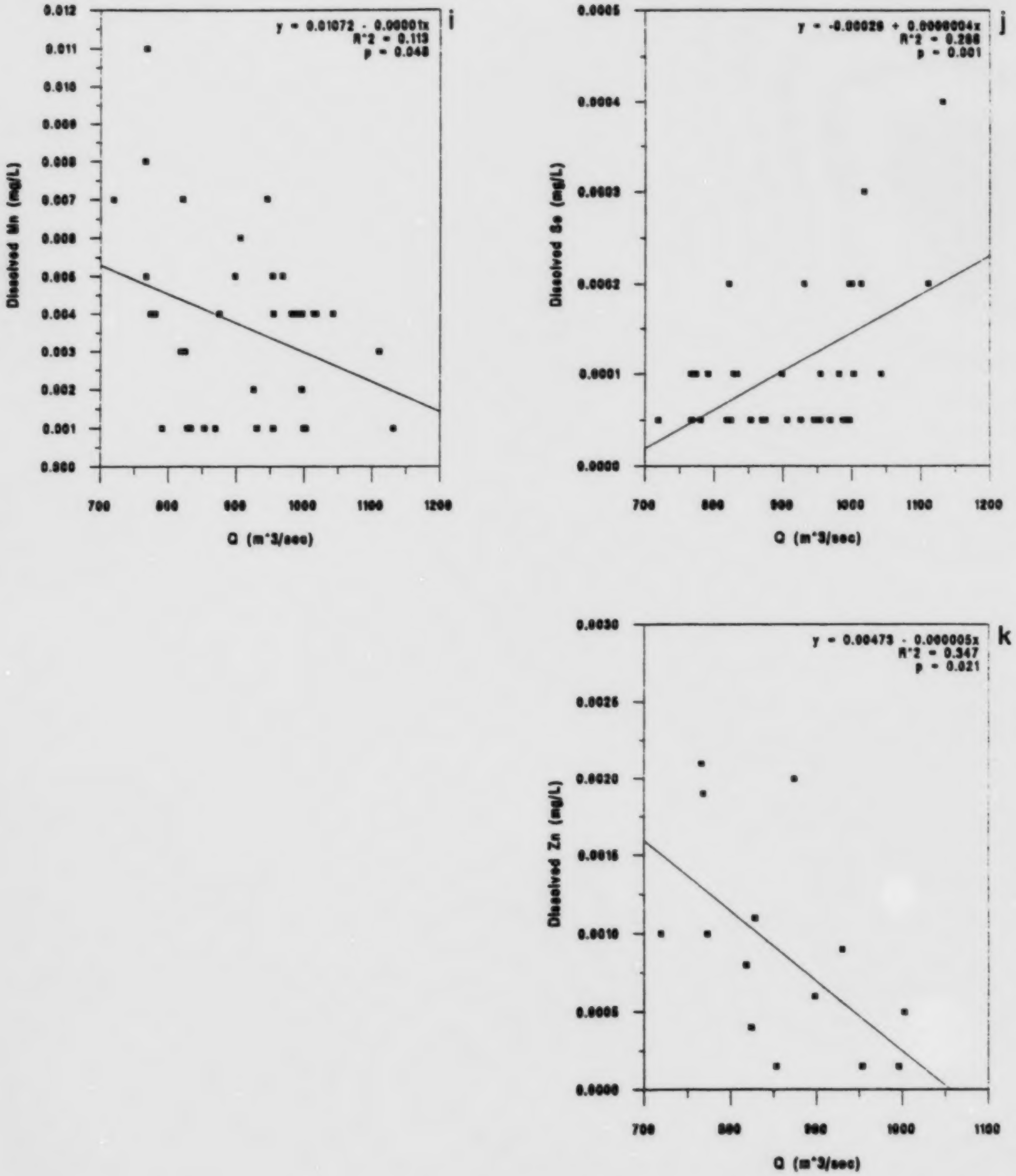

Figure 32. Continued.

No discharge analysis was possible for the pre-development period due to the lack of water quality data, although changes similar to those which occurred upstream at Thompson probably have occurred here as well.

3.5.1.3 Seasonal Differences

As found at a number of sites upstream on the Burntwood River, concentrations of dissolved NO_2-NO_3 were higher under ice-cover while chlorophyll a levels were lower under ice-cover than in the open-water season (Figure 33). Causes of these seasonal patterns are probably the same as at the upstream sites.

3.5.2 Nelson River below Grass River at Inlet to Split Lake

3.5.2.1 Current Water Quality

Physical and chemical water quality in the Nelson River at the inlet to Split Lake was not significantly different in any aspect from that at Sea River Falls, upstream (Table 23). As at all other sites in the study, mean turbidity exceeded the drinking water guideline as did peak colour (Table 24). Mean extractable Al and total Cu exceeded the respective aquatic life guidelines by a factor of 1.9 on average while the peak values were 4 and 8 times the guideline values. The maximum extractable Fe concentration was about 40% above the aquatic life and drinking water guidelines although the mean value was 36% below these limits.

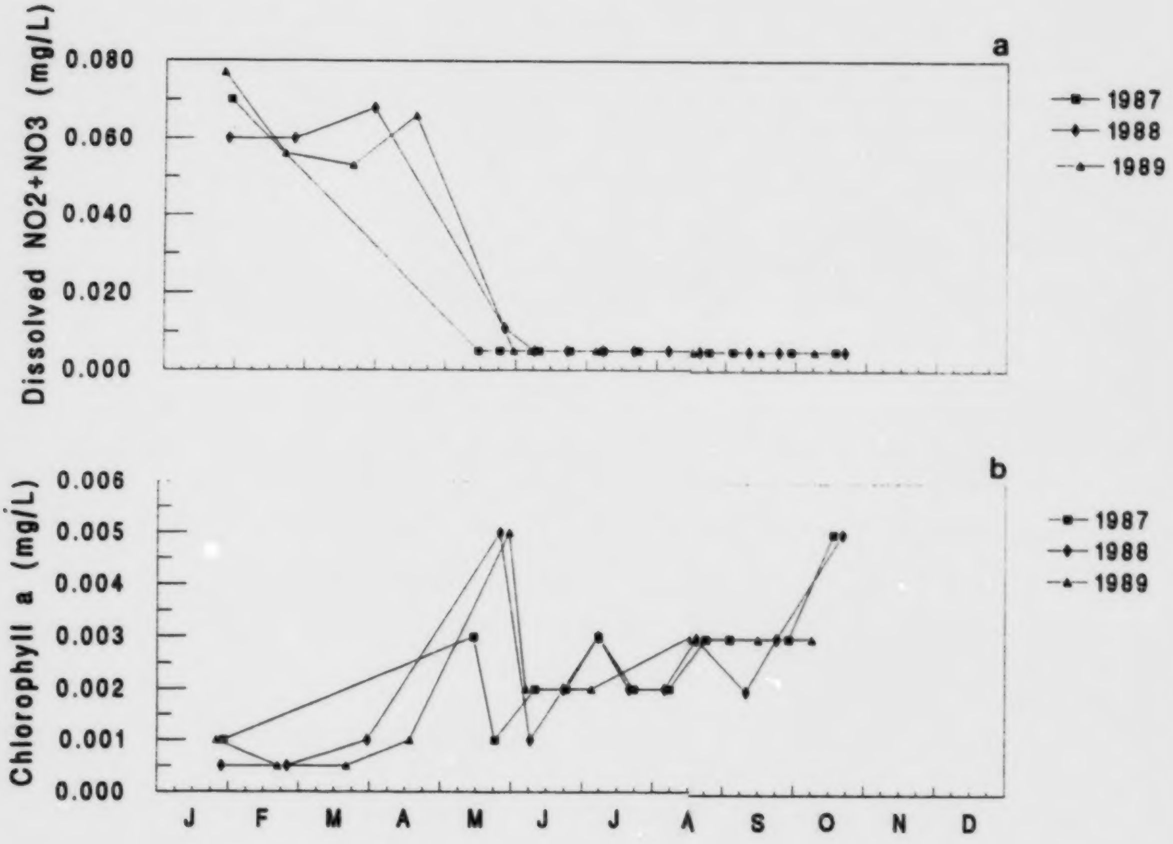

Figure 33. Seasonal water quality trends in the Burntwood River at the inlet to Split Lake. a. dissolved NO_2-NO_3 b. chlorophyll a.

Table 23. Comparison of current water quality in the Nelson River below Sea River Falls and at the inlet to Split Lake. X_{srf} denotes the 1987-1989 mean for the Nelson River below Sea River Falls. X_{spl} denotes the 1987-1989 mean for the Nelson River at the inlet to Split Lake. p is the probability of Student's t-statistic for the comparison.

Location		pH	Conductivity µS cm^{-1}	Turbidity NTU	Colour Relative	TDS mg L^{-1}	NFR mg L^{-1}	NFR Fixed mg L^{-1}	Tot Alk mg L^{-1}	Tot Hard mg L^{-1}	Diss Ca mg L^{-1}
Nelson R.	p	0.999	0.904	0.877	1.000	0.921	0.985	0.878	0.999	0.989	0.999
	X_{srf}	7.83^1	305.0^1	8.	9.	161.7^1	6.0^1	5.0^1	102.0^1	117.0^1	29.1^1
	X_{spl}	7.85	293.0	9.	9.	154.7	5.0	3.3	103.0	115.0	29.3

	Diss Mg mg L^{-1}	Diss K mg L^{-1}	Diss Na mg L^{-1}	React Si mg L^{-1}	Diss Cl mg L^{-1}	Diss F mg L^{-1}	Diss SO$_4$ mg L^{-1}	Ortho P mg L^{-1}	TDP mg L^{-1}	Tot P mg L^{-1}
p	0.855	0.610	0.753	0.475	0.774	0.982	0.832	0.097	0.946	0.940
X_{srf}	10.7^1	2.25^1	15.6^1	0.93	17.7^1	0.10	23.7^1	0.002	0.010	0.023
X_{spl}	10.2	2.06	13.8	1.24	15.3	0.10^1	21.1^1	0.004^1	0.011	0.022

	TDN mg L^{-1}	Diss NO$_2$-NO$_3$ mg L^{-1}	Susp N mg L^{-1}	DOC mg L^{-1}	TOC mg L^{-1}	DIC mg L^{-1}	Chlorophyll a mg L^{-1}
p	1.000	0.999	0.995	0.961	0.874	0.979	1.000
X_{srf}	0.301	0.014^1	0.051^1	7.67	8.05	24.8^1	0.002
X_{spl}	0.305	0.016	0.048	7.51	7.79	24.4	0.002

Table 23. continued.

Location		Extr Al mg L^{-1}	Diss Al mg L^{-1}	Diss As mg L^{-1}	Tot Cd mg L^{-1}	Diss Cd mg L^{-1}	Tot Co mg L^{-1}	Tot Cu mg L^{-1}	Diss Cu mg L^{-1}	Extr Fe mg L^{-1}	Diss Fe mg L^{-1}
Nelson R.	p	1.000	1.000	0.512	1.000	1.000	1.000	0.994	0.997	0.999	0.951
	X_{srf}	0.182	0.054	0.0006	0.0001	0.0001	0.0003	0.0050	0.0021	0.197[1]	0.033
	X_{spl}	0.186	0.055	0.0007	0.0001	0.0001	0.0003	0.0037	0.0020	0.193	0.039

	Tot Pb mg L^{-1}	Diss Pb mg L^{-1}	Extr Mn mg L^{-1}	Diss Mn mg L^{-1}	Tot Ni mg L^{-1}	Diss Ni mg L^{-1}	Diss Se mg L^{-1}	Tot V mg L^{-1}	Diss V mg L^{-1}	Tot Zn mg L^{-1}	Diss Zn mg L^{-1}
p	0.932	1.000	0.034	0.946	1.000	1.000	0.270	0.999	0.991	0.968	1.000
X_{srf}	0.0008	0.0009	0.016	0.004	0.0011	0.0009	0.0001	0.0008	0.0004	0.0033	0.0010
X_{spl}	0.0006	0.0010	0.010	0.003	0.0011	0.0009	0.0002	0.0008	0.0005	0.0025	0.0010

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[1] Discharge-standardized variables.

Table 24. Summary statistics for physical and chemical water quality in Nelson River below Grass River, September, 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.7	0.3	8.3	7.1
	Lab		7.85	0.21	8.16	7.30
Conductivity	Field	$\mu S\ cm^{-1}$	273	43	345	185
	Lab		291	38	360	232
Turbidity	Field	NTU	12^{+}	7	45^{+}	4
	Lab		9^{+}	3	18^{+}	5
Colour	True	Relative	9	5	20^{+}	<5
TDS		$mg\ L^{-1}$	154	21	195	119
NFR	Total	$mg\ L^{-1}$	5	2	11	2
	Fixed	$mg\ L^{-1}$	4	2	9	<1
Alkalinity	Total	$mg\ L^{-1}$	102.7	10.4	123.0	85.3
	Bicarbonate	$mg\ L^{-1}$	125.4	12.4	149.9	104.0
Hardness	Total	$mg\ L^{-1}$	114.4	13.4	141.9	91.4
	Non-carbonate	$mg\ L^{-1}$	11.9	4.6	21.1	3.9
Free CO_2		$mg\ L^{-1}$	4.25	3.52	15.28	0
Phosphorus	Particulate	$mg\ L^{-1}$	0.011	0.005	0.020	<0.006
	Ortho PO_4	$mg\ L^{-1}$	0.003	0.002	0.011	<0.003
	Total Dissolved	$mg\ L^{-1}$	0.011	0.003	0.020	0.005
	Total	$mg\ L^{-1}$	0.022	0.005	0.032	0.012
Nitrogen	Particulate	$mg\ L^{-1}$	0.048	0.059	0.380	<0.010
	Total Dissolved	$mg\ L^{-1}$	0.306	0.039	0.380	0.210
	NO_2-NO_3	$mg\ L^{-1}$	0.016	0.016	0.054	<0.010
	Total NH_3	$mg\ L^{-1}$	0.023	0.011	0.060	0.005
	Total	$mg\ L^{-1}$	0.35	0.04	0.43	0.24
Carbon	DOC	$mg\ L^{-1}$	7.55	0.50	8.70	6.34
	POC	$mg\ L^{-1}$	0.285	0.082	0.520	0.140
	TOC	$mg\ L^{-1}$	7.83	0.49	8.86	6.64
	DIC	$mg\ L^{-1}$	24.3	2.7	29.5	19.8
Chlorophyll-a		$mg\ L^{-1}$	0.002	0.002	0.010	<0.001
Ca	Dissolved	$mg\ L^{-1}$	29.2	3.1	35.4	23.5
Mg	Dissolved	$mg\ L^{-1}$	10.1	1.4	13.0	7.8
K	Dissolved	$mg\ L^{-1}$	2.03	0.29	2.64	1.61
Na	Dissolved	$mg\ L^{-1}$	13.6	2.9	19.1	9.2

Table 24. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	1.29	0.66	2.94	0.34
Cl	Dissolved	mg L^{-1}	14.9	3.8	22.5	9.2
F	Dissolved	mg L^{-1}	0.10	0.01	0.12	0.08
SO_4	Dissolved	mg L^{-1}	20.7	4.6	30.0	13.1
Al	Extractable	mg L^{-1}	0.188^{+++}	0.101	0.418^{+++}	<0.100
	Dissolved	mg L^{-1}	<0.100	0.016	0.119	<0.100
As	Dissolved	mg L^{-1}	0.0007	0.0002	0.0010	0.0004
Ba	Total	mg L^{-1}	<0.08		<0.08	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L^{-1}	<0.0001		0.0002	<0.0001
Co	Total	mg L^{-1}	<0.0005		0.0005	<0.0005
Cu	Total	mg L^{-1}	0.0037^{+++}	0.0043	0.0159^{+++}	0.0010
	Dissolved	mg L^{-1}	0.0020	0.0007	0.0031	0.0010
Fe	Extractable	mg L^{-1}	0.196	0.074	0.409^{+++}	0.045
	Dissolved	mg L^{-1}	0.043	0.030	0.178	0.013
Pb	Total	mg L^{-1}	<0.0007	0.0004	0.0017	<0.0007
	Dissolved	mg L^{-1}	0.0011	0.0020	0.0072	<0.0007
Mn	Extractable	mg L^{-1}	0.010	0.003	0.020	0.004
	Dissolved	mg L^{-1}	0.003	0.003	0.018	<0.002
Ni	Total	mg L^{-1}	0.0011	0.0004	0.0023	0.0005
	Dissolved	mg L^{-1}	0.0009	0.0004	0.0017	<0.0005
Se	Dissolved	mg L^{-1}	0.0002	0.0001	0.0005	<0.0001
V	Total	mg L^{-1}	0.0008	0.0004	0.0014	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0002	0.0010	<0.0005
Zn	Total	mg L^{-1}	0.0024	0.0020	0.0077	0.0006
	Dissolved	mg L^{-1}	0.0010	0.0009	0.0027	<0.0003
Discharge	Daily	m^3 sec^{-1}	1271	352	2253	734

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recrearional use.

$^{+++}$ Exceeds limit for protection of aquatic life.

3.5.2.2 Water Chemistry - Water Discharge Relationships

3.5.2.2.1 Pre-development

Pre-development data from 3 sites were used in the analysis, with the discharge analysis conducted separately on each data set. At the Nelson River below Sipiwesk Lake, the only significant correlation was a positive relationship with NFR, which accounted for 67% of the variance (Figure 34). At the Nelson River above Kelsey Generating Station, 9 variables were correlated with discharge. Negative correlates included conductivity (field), dissolved Na, K, Mg, Ca, SO_4, TDS, and dissolved Fe (Figure 35). The sole positive correlate was POC. All of the regressions accounted for at least 45% of the variance in the dependent variables. The correlations of discharge with Ca, K, SO_4, and conductivity were particularly strong, accounting for 78 to 93% of the variance.

At the Nelson River below Kelsey Generating Station, 4 variables were significantly correlated with discharge (Figure 36). Positive correlates were NFR and TDN while the negative correlates were conductivity (field) and TDS. The regressions generally were not as strong as found above the Kelsey Dam, accounting for 30 to 66% of the variance in the dependent variables.

There are some general similarities in the discharge relationships at the three stations. Conductivity and TDS were both negatively correlated with discharge both above and below Kelsey, while NFR was positively correlated with discharge below Sipiwesk Lake and below Kelsey. Little significance should be attached to

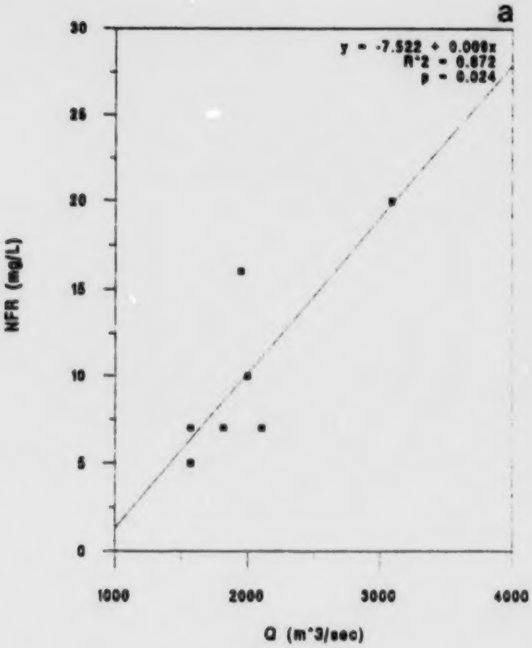

Figure 34. Water chemistry - water discharge relationships in the Nelson River below Sipiwesk Lake, 1972-1973. a. NFR.

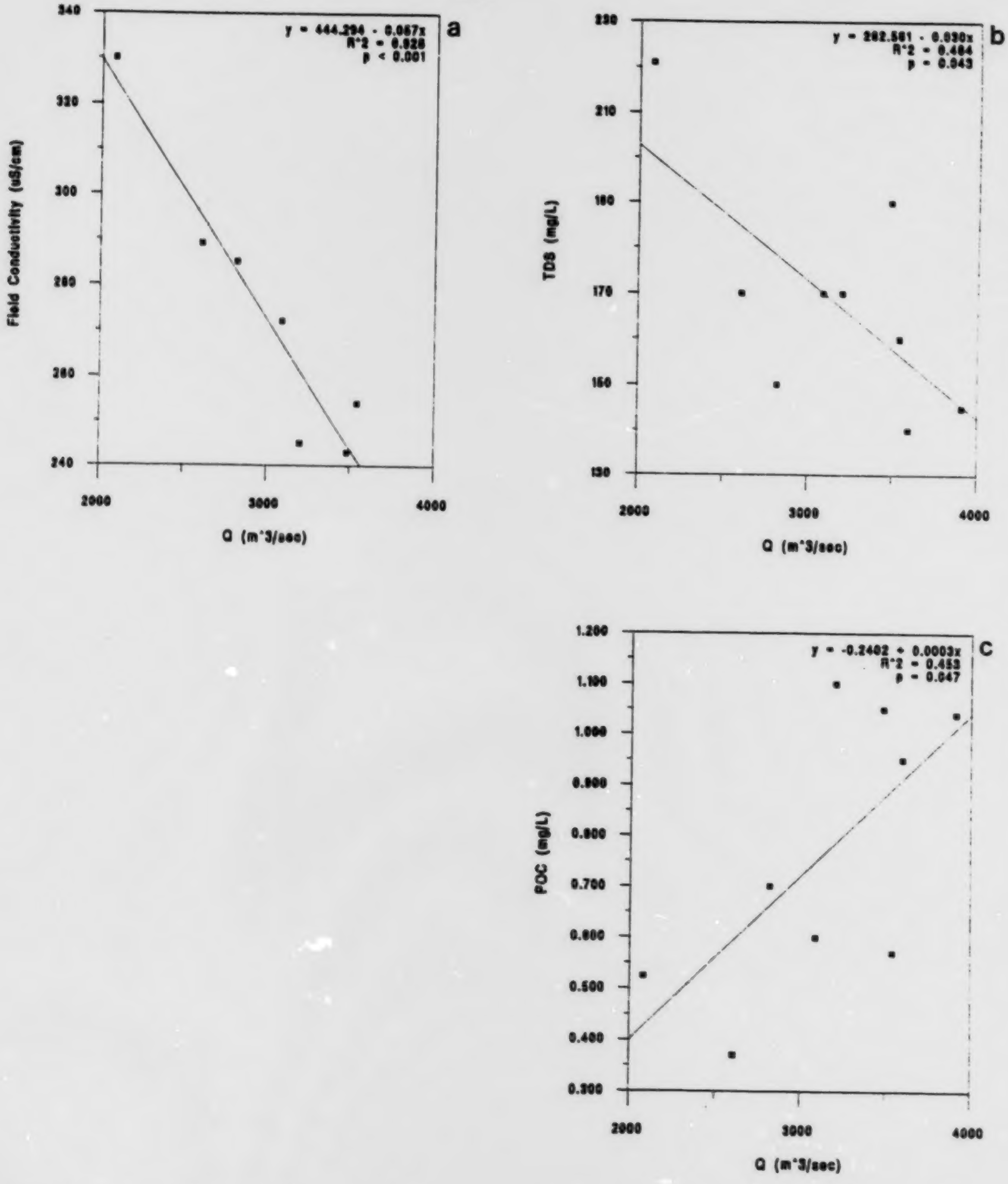

Figure 35. Water chemistry - water discharge relationships in the Nelson River above Kelsey Generating Station, 1972-1974. a. field conductivity b. TDS c. POC d. dissolved Na e. dissolved SO_4 f. dissolved Fe g. dissolved Ca H. dissolved Mg i. dissolved K.

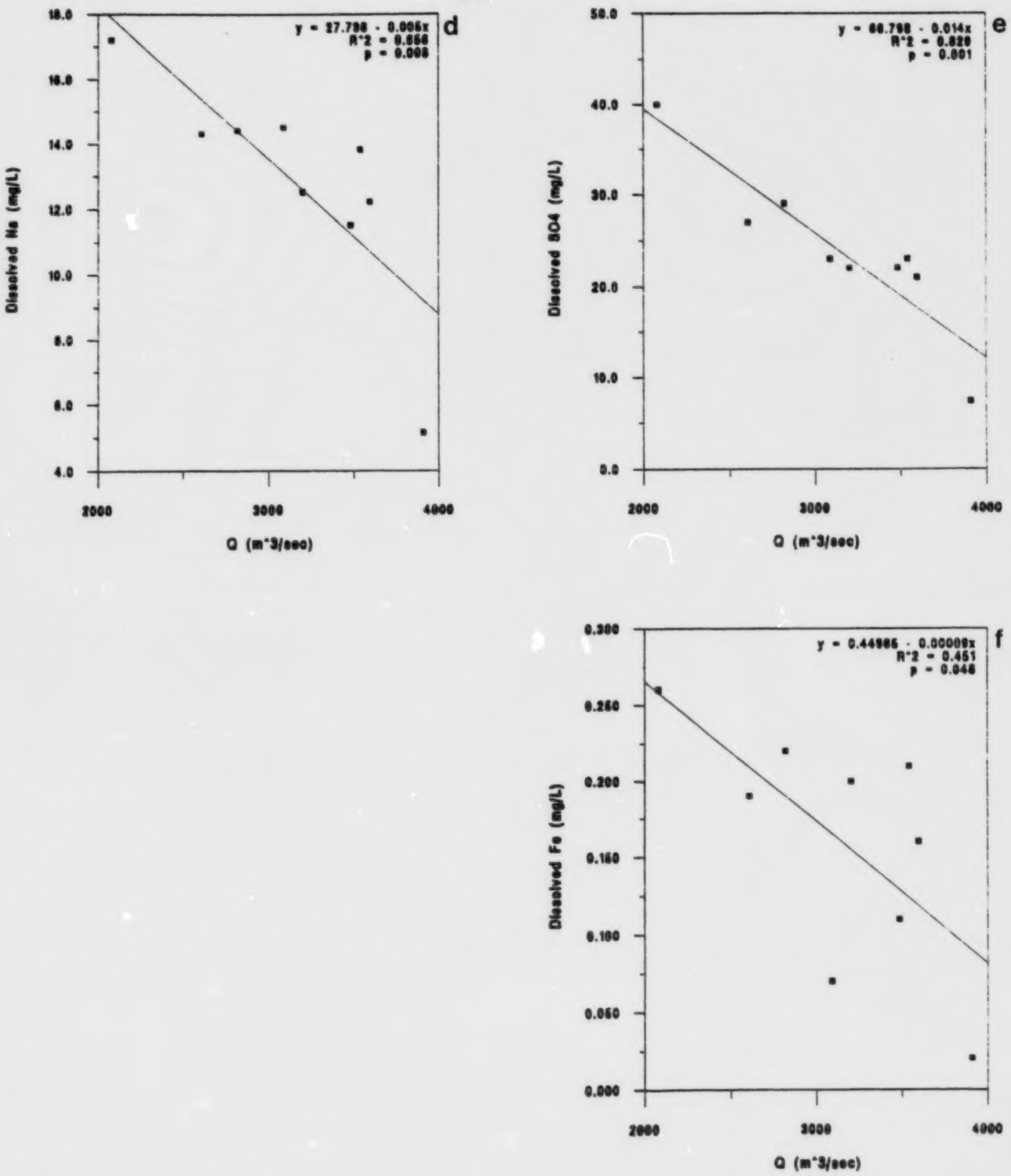

Figure 35. Continued.

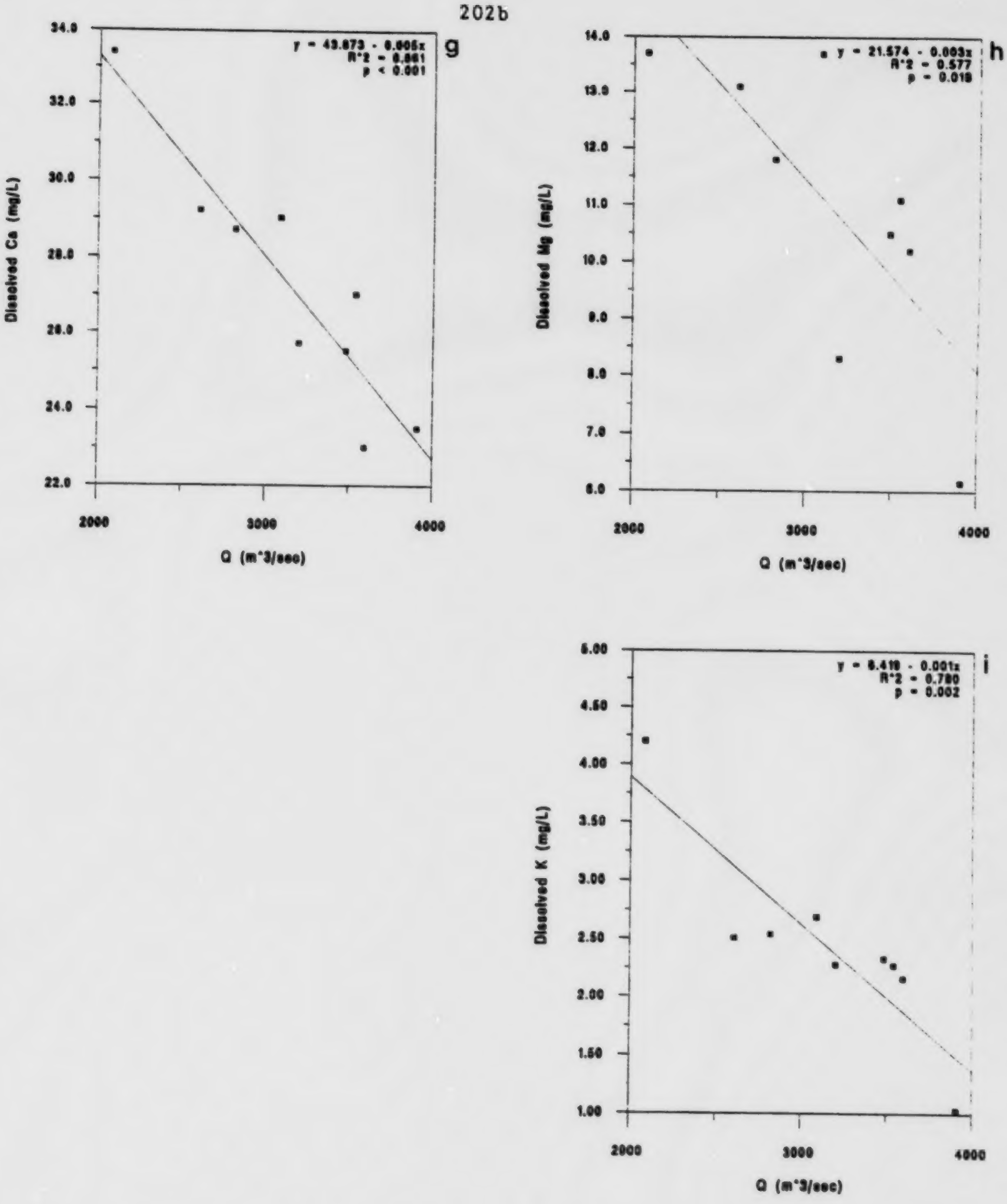

Figure 35. Continued.

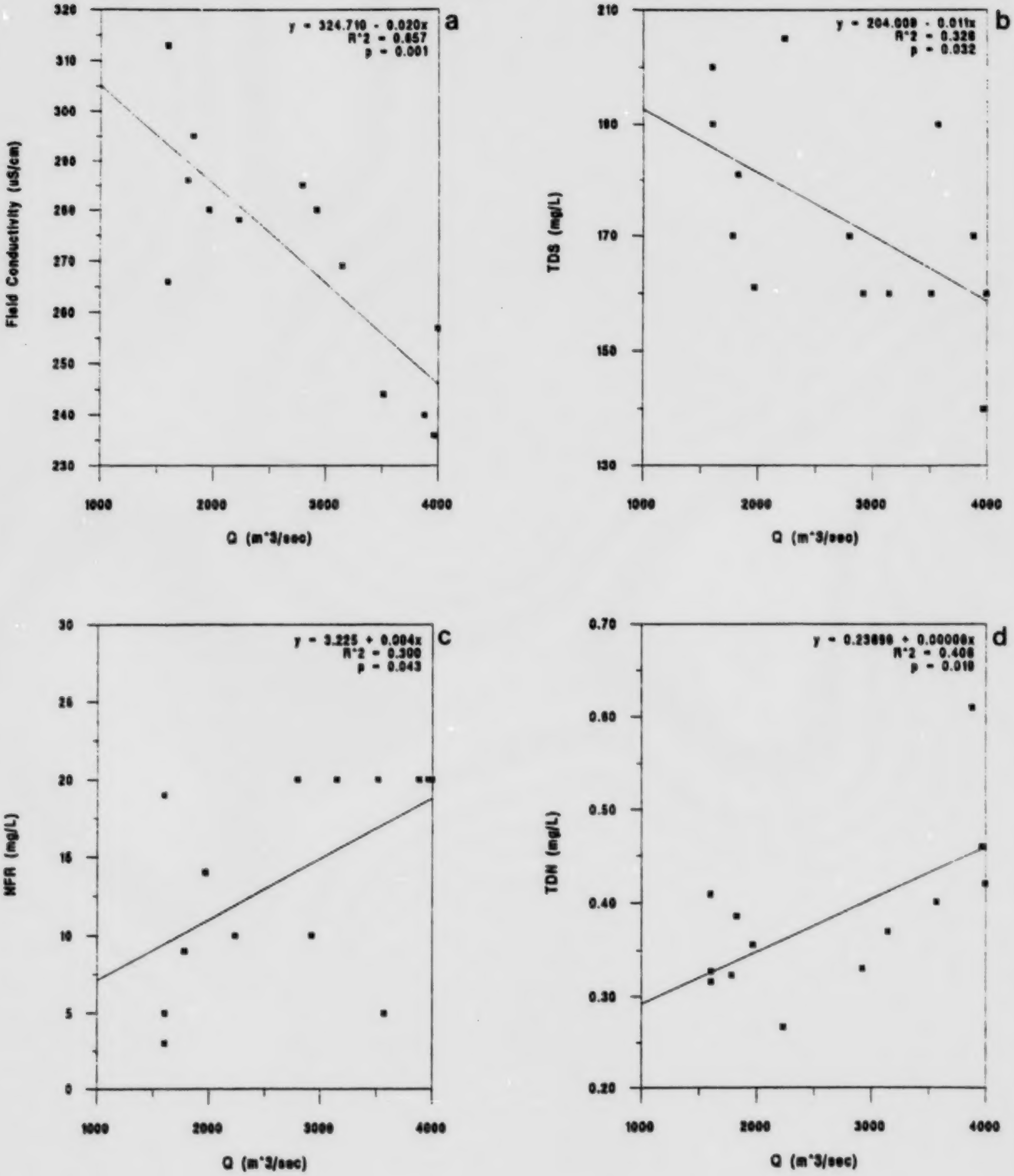

Figure 36. Water chemistry - water discharge relationships in the Nelson River below Kelsey Generating Station, 1972-1974. a. field conductivity b. TDS c. NFR d. TDN.

any specific differences in these common relationships, however, as all are based on few samples, the samples were not collected on the same date at all stations and, therefore, the range of discharge on which the relationships are based was not the same at each station.

The pre-development discharge relationships at this site also follow the trends evident in the natural rivers and in the Burntwood River at Thompson prior to development. Negative correlates predominated with positive correlations restricted to variables associated with particulate matter or suspended sediments.

3.5.2.2.2 Current

Fewer variables were significantly correlated with discharge at this site than at sites in the Burntwood River drainage or upstream on the Nelson River. Most of the significant correlations had positive slopes. These included ortho PO_4, dissolved fluoride, SO_4, and Cd, and total Cd (Figure 37). The only negative correlate was total N. None of these correlations was particularly strong, with all but one accounting for just 10-20% of the variance in the dependent variable.

None of the significant correlations with discharge in the current data resembles any which existed during the pre-development (1972-1973) period, but the cause of these changes is less clear than was the case at sites on the Burntwood River. The changes are superficially similar to those caused by CRD (i.e., positive slopes for variables which might otherwise be expected to be negative

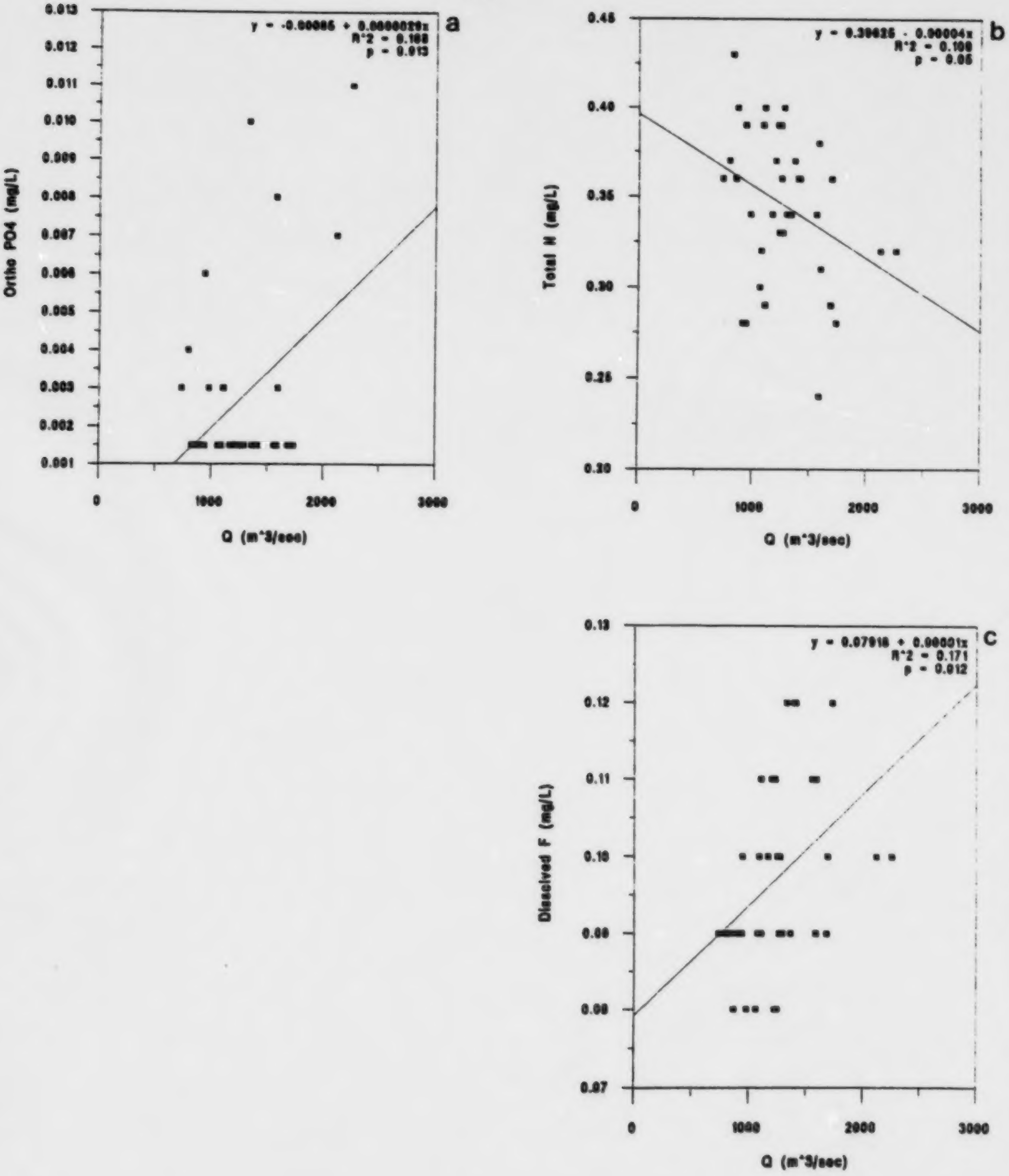

Figure 37. Water chemistry - water discharge relationships in the Nelson River at the inlet to Split Lake, 1987-1989. a. ortho PO_4 b. total N c. dissolved F d. dissolved SO_4 e. dissolved Cd f. total Cd.

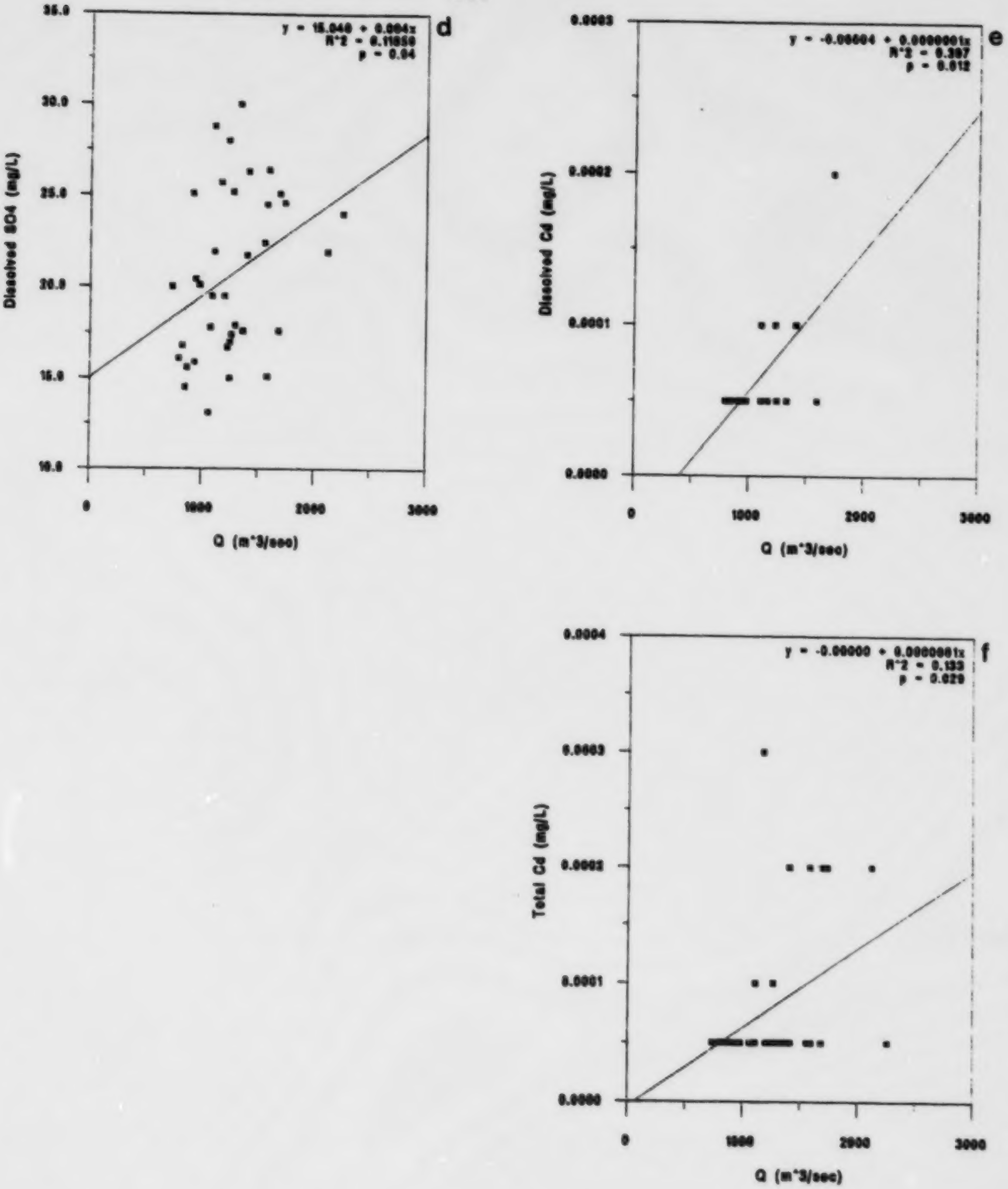

Figure 37. Continued.

correlates of discharge). Alternatively, these changes may be a result of the low discharges and rather narrow range of discharge fluctuation during the FEMP study compared with the pre-development period, resulting in little overlap in the discharges which were common to the two periods. It is questionable if any of the correlations observed in the FEMP data would prevail over a broader range of discharges. On this basis, it cannot be determined if LWR has had any effect on the water quality - water discharge relationships at this site. Additional sampling during periods of higher discharge is required to determine if the relationships observed during the FEMP study are representative of current conditions or if some other relationships represent the usual post-development condition. The discharge relationships derived from the FEMP data are not recommended for use in prediction or management without additional data collection over a wider range of discharges.

3.5.2.3 Seasonal Differences

Few variables showed marked differences between the open-water and ice-cover seasons during the 1987-1989 FEMP sampling period although conductivity, hardness (total), dissolved Mg, and dissolved NO_2-NO_3 tended to be higher under ice cover (Figure 38). The cause of the higher NO_2-NO_3 under ice is probably the same as discussed in Section 3.3.1.3. The elevated hardness is a result of the higher Mg, but the cause of the elevated Mg is unclear. Mg was not correlated with discharge at this site and, in general, is not

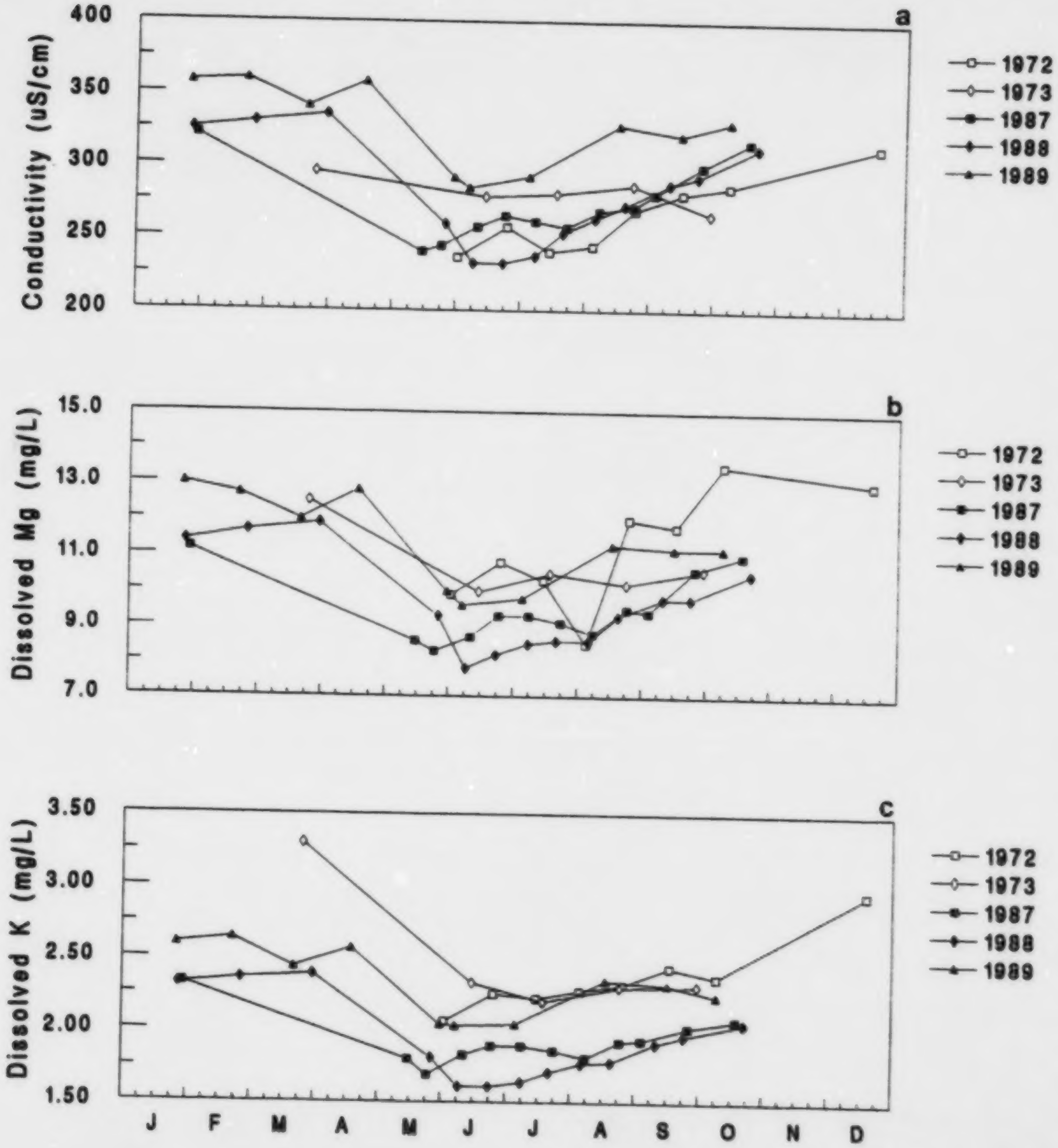

Figure 38. Seasonal water quality trends in the Nelson River at the inlet to Split Lake. a. lab conductivity b. dissolved Mg c. dissolved K d. total hardness e. dissolved NO_2-NO_3 f. DIC.

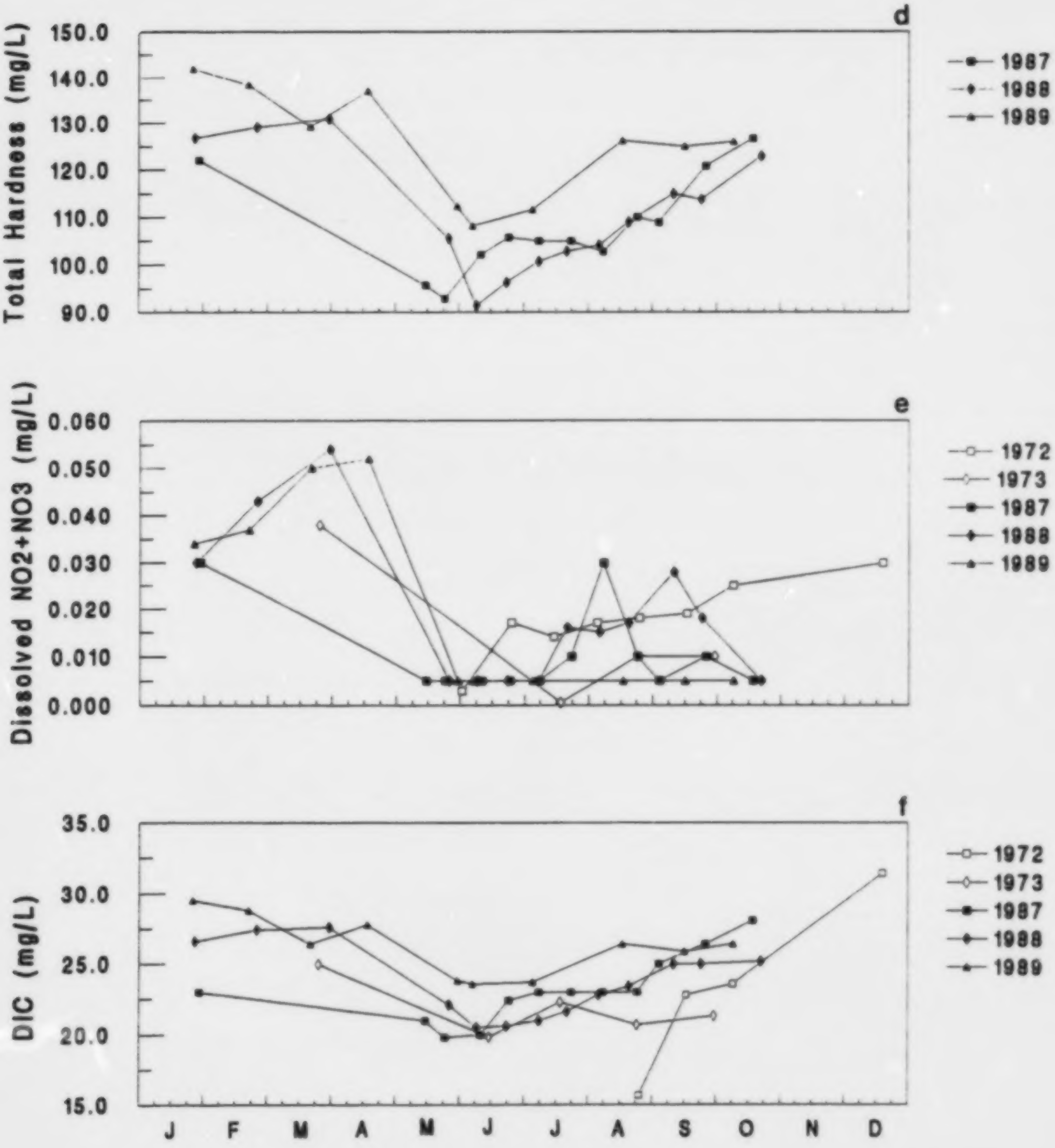

Figure 38. Continued.

known for marked seasonal variations due to its considerable solubility and minor biotic demand (Wetzel 1983). Conductivity was not correlated with discharge so its seasonality also cannot be explained by discharge variations.

The pre-development database for this site is poor for the examination of differences between the ice-free and ice-cover seasons. Only one winter measurement was taken from each of the Nelson River above Kelsey Dam (Mar. 1973) and the Nelson River below Sipiwesk Lake (Feb. 1973), while 2 were taken from the Nelson River below Kelsey Dam (Dec. 1972 and Mar. 1973). These limited data preclude a statistical analysis of changes in seasonal water quality patterns. Qualitative analysis of the data from the Nelson River below Kelsey Dam reveals higher conductivity, dissolved K and Mg, DIC, and NO_3 under ice-cover than during the open water season (Figure 38). Conductivity and dissolved K and Mg all were negatively correlated with discharge prior to development (Figure 36), with the lower winter discharges a possible explanation for the increases under ice cover. Again, the increased NO_3-N under ice-cover probably was a product of reduced photosynthetic demand. The same process may explain the increase in DIC under ice. DIC is the inorganic carbon source for photosynthetic organisms (Wetzel 1983), but this explanation seems improbable given the relatively high buffering capacity of the Nelson River waters.

Although it appears the seasonal patterns of several variables were somewhat different during the FEMP study than prior to LWR, it would be inappropriate to attribute these changes to the hydroelectric development. The low Nelson River flows during the FEMP study may have been responsible. The pre-development seasonal trends were based on a single winter and a minimum number of observations so it is not known if these apparent changes exceed the range of natural variation. The influence of LWR on seasonal trends in water quality should be clarified through additional monitoring during periods of higher flow.

3.5.2.4 Changes in Water Quality Since Lake Winnipeg Regulation

Significant changes occurred in 6 water quality variables between the 1972-1973 pre-development study period and the 1987-1989 FEMP study (Table 25). pH declined by 5%. NFR, dissolved K, TDP, and TDN also declined by 67, 15, 45, and 22%, respectively. The reactive Si concentration increased by 72%. A marginally significant decline (p=0.075) in chlorophyll a, by 51%, also occurred.

There is no clear relationship between these changes in water quality and LWR. The decline in pH was comparable to that observed in the natural rivers and is probably due to changes in analytical equipment or procedure. All of the other water quality changes appear to have been restricted to the reach between Sipiwesk and Split lakes. No significant changes in any of the nitrogen or phosphorus species, reactive Si, major ions (including dissolved K), turbidity, NFR, TDS, conductivity, suspended carbon, or colour on the Nelson River mainstem in Sipiwesk Lake were found in a comparison of baseline data from 1972-1973 with data for the period

Table 25. Evaluation of the significance of changes in selected water quality parameters in the Nelson River near the inlet of Split Lake following Lake Winnipeg regulation. X_{pre} denotes the mean for pre-development years, 1972 and 1973. X_{post} denotes the mean for the post-development years, 1987 through 1989. p is the probability of Student's t-statistic for the comparison.

Location		pH	Conductivity µS cm^{-1}	TDS mg L^{-1}	NFR NTU	Diss Ca mg L^{-1}	Diss Mg mg L^{-1}	Diss K mg L^{-1}	Diss Na mg L^{-1}	React Si mg L^{-1}	Diss Cl mg L^{-1}
Nelson R.	p	0.003	0.904	0.121	0.018	0.570	0.182	0.033	0.838	0.025	0.330
(Split L inlet)	X_{pre}	8.24	273.0^{2,3}	177.0^{2,3}	15.0^{1,3}	28.5^{2}	11.1^{2}	2.43^{2}	14.1^{2}	0.72	17.3
	X_{post}	7.85	275.0	155.0	5.0	29.3	10.2	2.06	13.8	1.24	15.3

		Diss SO$_4^{-2}$ mg L^{-1}	TDP mg L^{-1}	Susp P mg L^{-1}	TDN mg L^{-1}	Susp N mg L^{-1}	DOC mg L^{-1}	DIC mg L^{-1}	Chlorophyll a mg L^{-1}	Diss Fe mg L^{-1}	Diss Mn mg L^{-1}
	p	0.162	0.017	0.42	0.003	0.120	0.879	0.123	0.075	0.179	0.044
	X_{pre}	24.5^{2}	0.020	0.039	0.388^{3}	0.068	7.39	22.2	4.700	0.113^{2}	0.039
	X_{post}	21.1^{1}	0.011	0.011	0.304	0.048	7.51	24.3	2.333	0.039	0.003

X_{pre} [1] Station 37a

[2] Station 38

[3] Station 39

X_{post} [1] Discharge-standardized variables.

from 1987 to 1989 (Berger and Ramsey 1991). There were no significant changes in NFR, reactive Si, or major ion concentrations in Cross Lake during the same period (Berger and Ramsey 1991). Significant changes in several parameters were reported by Playle and Williamson (1986) for the Nelson River at Norway House, Cross Lake, and Sipiwesk Lake, although none can account for the observed changes in the Nelson River at the inlet to Split Lake.

Some of these changes might be explained by a natural evolution of limnological conditions in the Kelsey Generating Station forebay. Severe shoreline erosion became evident in the forebay after flooding (Water Resources Branch 1974), and the observed reduction in NFR might be an indication these shorelines are beginning to stabilize. Alternatively, erosion rates may have decreased temporarily as a result of the lower Nelson River flows. Between 1987 and 1989, Nelson River flows were at or near the lowest observed in the past 30 years (Figure 2). Based on the correlation between NFR and discharge in the Nelson River below Sipiwesk Lake for the 1972-1973 sampling period (Figure 34), mean NFR at the mean 1987-1989 discharge (1271 m^3 sec^{-1}) is estimated to be 5 mg L^{-1}, compared with 15 mg L^{-1} at the mean 1972-1973 discharge (2305 m^3 sec^{-1}). The observed reduction in NFR is identical (Table 25) and therefore appears to be completely accounted for by the reduction in discharge. Neither of these explanations can be ruled out with the data at hand. Additional sampling after the current drought conditions have subsided is required to properly determine if LWR has had any long-term impact on water quality at this site.

3.5.3 Differences in Nelson and Burntwood River Water Quality

Water quality in the Nelson and Burntwood rivers differed significantly in many aspects in the 1987-1989 study period. Concentrations of most major ions and values of related variables (i.e., conductivity, alkalinity, DIC, hardness, and TDS) were significantly lower in the Burntwood River (Table 26). Most values were 45-60% lower, although Na, Cl, and SO_4 concentrations were 80-92% lower.

The Burntwood River was significantly more turbid, by about a factor of 4, and carried an almost 5-fold higher suspended sediment (NFR) concentration than the Nelson River (Table 26). Several metal species also occurred in significantly higher concentrations in the Burntwood River. Levels of extractable Al, Fe, and Mn were 3 to 4 times higher while total Ni and V levels were a factor of 2 higher. A similar trend was evident in the dissolved metals, although the difference was not as large and fewer metals were affected. The mean dissolved Al concentration was 50% higher in the Burntwood River and dissolved Fe was 2.3 times higher. In contrast to the metals, concentrations of dissolved As (a semi-metal) and dissolved Se (a metalloid) in the Burntwood River were 71 and 50% of levels in the Nelson River.

No clear trend was evident among the nutrient species (Table 26). Concentrations of reactive Si and total P were significantly higher, by 58 and 77%, respectively, in the Burntwood River while

Table 26. Comparison of current water quality in the Nelson and Burntwood rivers at the inlet to Split Lake. X_{nel} denotes the 1987-1989 mean for the Nelson River. X_{bwr} denotes the 1987-1989 mean for the Burntwood River. p is the probability of Student's t-statistic for the comparison.

Location		pH	Conductivity $\mu S\ cm^{-1}$	Turbidity NTU	Colour Relative	TDS $mg\ L^{-1}$	NFR $mg\ L^{-1}$	NFR Fixed $mg\ L^{-1}$	Tot Alk $mg\ L^{-1}$	Tot Hard $mg\ L^{-1}$	Diss Ca $mg\ L^{-1}$
Split Lake	p	0.896	<0.001	<0.001	0.006	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
(inlet)	X_{nel}	7.85	293.	9.0	9.	154.7	5.0	3.3	103.0	115.0	29.3
	X_{bwr}	7.76	123.	35.0^1	22.	64.7	24.3	21.3	56.7^1	56.7	15.9
		Diss Mg $mg\ L^{-1}$	Diss K $mg\ L^{-1}$	Diss Na $mg\ L^{-1}$	React Si $mg\ L^{-1}$	Diss Cl $mg\ L^{-1}$	Diss F $mg\ L^{-1}$	Diss SO_4 $mg\ L^{-1}$	Ortho P $mg\ L^{-1}$	TDP $mg\ L^{-1}$	Tot P $mg\ L^{-1}$
	p	<0.001	<0.001	<0.001	0.020	<0.001	0.982	<0.001	0.624	0.946	<0.001
	X_{nel}	10.2	2.06	13.8	1.24	15.3	0.10^1	21.1^1	0.004^1	0.011	0.022
	X_{bwr}	4.1	1.05	2.8	1.96	1.2	0.10	2.9	0.003	0.012	0.039
					TDN $mg\ L^{-1}$	Diss NO_2-NO_3 $mg\ L^{-1}$	Susp N $mg\ L^{-1}$	DOC $mg\ L^{-1}$	TOC $mg\ L^{-1}$	DIC $mg\ L^{-1}$	Chlorophyll a $mg\ L^{-1}$
	p				0.053	0.998	0.438	0.062	0.587	<0.001	1.000
	X_{nel}				0.305	0.016	0.048	7.51	7.79	24.4	0.002
	X_{bwr}				0.248	0.019	0.063	6.78	7.41^1	13.4	0.002

Table 26. continued.

Location		Extr Al mg L^{-1}	Diss Al mg L^{-1}	Diss As mg L^{-1}	Tot Cd mg L^{-1}	Diss Cd mg L^{-1}	Tot Co mg L^{-1}	Tot Cu mg L^{-1}	Diss Cu mg L^{-1}	Extr Fe mg L^{-1}	Diss Fe mg L^{-1}
Split Lake	p	<0.001	0.006	<0.001	1.000	1.000	0.189	0.998	0.451	<0.001	0.001
(inlet)	X_{nat}	0.186	0.055	0.0007	0.0001[1]	0.0001	0.0003	0.0037	0.0020	0.193	0.039
	X_{twr}	0.596	0.082	0.0002	0.0001	0.0001	0.0005	0.0047	0.0015	0.787	0.091

		Tot Pb mg L^{-1}	Diss Pb mg L^{-1}	Extr Mn mg L^{-1}	Diss Mn mg L^{-1}	Tot Ni mg L^{-1}	Diss Ni mg L^{-1}	Diss Se mg L^{-1}	Tot V mg L^{-1}	Diss V mg L^{-1}	Tot Zn mg L^{-1}	Diss Zn mg L^{-1}
	p	0.484	0.996	<0.001	0.946	0.003	0.675	0.053	0.002	0.991	0.45	0.978
	X_{nat}	0.0006	0.0010	0.010	0.003	0.0011	0.0009	0.0002	0.0008	0.0005	0.0025	0.0010
	X_{twr}	0.0011	0.0007	0.031	0.004	0.0023	0.0011	0.0001[1]	0.0016	0.0004	0.0046	0.0007[1]

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[1] Discharge-standardized variables.

TDN and DOC were marginally higher, by 23 and 11%, in the Nelson River. The higher total P concentration in the Burntwood River is entirely attributable to the suspended fraction as there were no differences in TDP or ortho PO_4 between the rivers.

3.5.4 Nelson River at Outlet of Split Lake

3.5.4.1 Current Water Quality

All physical and chemical water quality data gathered at this site during the FEMP study are summarized in Table 27. Water quality at the outlet of Split Lake shared a number of general similarities with sites upstream on the Nelson River. Mean turbidity and maximum colour exceeded the drinking water guidelines and the mean extractable Al and total Cu exceeded the guidelines for protection of aquatic life. However, water quality here differed from the other sites on the Nelson River in the magnitude by which these guidelines were exceeded. Turbidity exceeded the guideline by a factor of 2 on average at the other Nelson River sites, compared with a factor of 4 at the outlet of Split Lake. Mean extractable Al also was about a factor of 2 above the guideline value at the upstream sites compared with a factor of 3.2 at the outlet.

Other differences between the Split Lake outlet and sites on the upper Nelson River include an extractable Fe concentration exceeding the drinking water and aquatic life guidelines by about 30% on average and by as much as a factor of 2. Upstream, peak extractable Fe concentrations never were more than 50% over the

Table 27. Summary statistics for physical and chemical water quality in the Nelson River at the outlet of Split Lake, September 1986, to October, 1989. s.d. is standard deviation.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
pH	Field		7.7	0.3	8.3	7.1
	Lab		7.80	0.21	8.15	7.17
Conductivity	Field	µS cm^{-1}	212	33	278	130
	Lab		227	30	296	180
Turbidity	Field	NTU	22^{+}	12	$88^{+,++}$	13^{+}
	Lab		18^{+}	5	35^{+}	11^{+}
Colour	True	Relative	14	6	30^{+}	5
TDS		mg L^{-1}	118	16	154	93
NFR	Total	mg L^{-1}	10	4	21	4
	Fixed	mg L^{-1}	8	3	19	3
Alkalinity	Total	mg L^{-1}	84.1	8.3	101.0	70.0
	Bicarbonate	mg L^{-1}	102.5	10.2	123.1	85.3
Hardness	Total	mg L^{-1}	91.4	10.3	114.6	75.4
	Non-carbonate	mg L^{-1}	7.3	3.5	18.4	0
Free CO_2		mg L^{-1}	3.63	3.15	12.73	0
Phosphorus	Particulate	mg L^{-1}	0.017	0.006	0.030	<0.006
	Ortho PO_4	mg L^{-1}	0.003	0.003	0.015	<0.003
	Total Dissolved	mg L^{-1}	0.011	0.004	0.020	0.003
	Total	mg L^{-1}	0.028	0.005	0.040	0.020
Nitrogen	Particulate	mg L^{-1}	0.052	0.023	0.110	<0.010
	Total Dissolved	mg L^{-1}	0.296	0.078	0.660	0.210
	NO_2-NO_3	mg L^{-1}	0.030	0.074	0.404	<0.010
	Total NH_3	mg L^{-1}	0.016	0.009	0.047	0.005
	Total	mg L^{-1}	0.35	0.08	0.71	0.24
Carbon	DOC	mg L^{-1}	7.27	0.45	8.70	6.50
	POC	mg L^{-1}	0.428	0.102	0.720	0.250
	TOC	mg L^{-1}	7.72	0.47	8.95	6.93
	DIC	mg L^{-1}	19.8	2.1	24.2	15.7
Chlorophyll-a		mg L^{-1}	0.003	0.002	0.007	<0.001
Ca	Dissolved	mg L^{-1}	24.0	2.4	29.0	20.3
Mg	Dissolved	mg L^{-1}	7.7	1.1	10.3	6.0
K	Dissolved	mg L^{-1}	1.63	0.22	2.20	1.22
Na	Dissolved	mg L^{-1}	9.2	2.1	14.5	5.8

Table 27. continued.

Variable	Type	Units	Mean	s.d.	Maximum	Minimum
Si	Reactive	mg L^{-1}	1.54	0.53	2.60	0.69
Cl	Dissolved	mg L^{-1}	9.2	2.8	16.7	4.8
F	Dissolved	mg L^{-1}	0.10	0.02	0.15	0.08
SO_4	Dissolved	mg L^{-1}	13.5	3.4	21.3	8.6
Al	Extractable	mg L^{-1}	0.315[+++]	0.126	0.651[+++]	<0.100
	Dissolved	mg L^{-1}	<0.100	0.025	0.138	<0.100
As	Dissolved	mg L^{-1}	0.0005	0.0002	0.0009	0.0002
Ba	Total	mg L^{-1}	<0.08		<0.08	<0.08
Cd	Total	mg L^{-1}	<0.0001	0.0001	0.0003	<0.0001
	Dissolved	mg L^{-1}	<0.0001	0.0001	0.0003	<0.0001
Co	Total	mg L^{-1}	<0.0005	0.0001	0.0007	<0.0005
Cu	Total	mg L^{-1}	0.0050[+++]	0.0075	0.0308[+++]	0.0010
	Dissolved	mg L^{-1}	0.0020	0.0010	0.0042	0.0011
Fe	Extractable	mg L^{-1}	0.389[+,+++]	0.110	0.672[+,+++]	0.129
	Dissolved	mg L^{-1}	0.070	0.055	0.355	0.023
Pb	Total	mg L^{-1}	0.0009	0.0011	0.0066[+++]	<0.0007
	Dissolved	mg L^{-1}	0.0012	0.0015	0.0049	<0.0007
Mn	Extractable	mg L^{-1}	0.015	0.004	0.025	0.003
	Dissolved	mg L^{-1}	0.004	0.004	0.021	<0.002
Ni	Total	mg L^{-1}	0.0015	0.0004	0.0025	0.0006
	Dissolved	mg L^{-1}	0.0011	0.0004	0.0021	0.0006
Se	Dissolved	mg L^{-1}	0.0001	0.0001	0.0005	0
V	Total	mg L^{-1}	0.0011	0.0005	0.0020	<0.0005
	Dissolved	mg L^{-1}	<0.0005	0.0003	0.0009	<0.0005
Zn	Total	mg L^{-1}	0.0040	0.0049	0.0215	0.0006
	Dissolved	mg L^{-1}	0.0013	0.0018	0.0055	<0.0003
Discharge	Daily	m^3 sec^{-1}	2641	577	3817	1530

[+] Exceeds limit for drinking water.

[++] Exceeds limit for recreational use.

[+++] Exceeds limit for protection of aquatic life.

water quality objectives. Similarly, total Pb levels never exceeded the water quality guidelines at upstream sites yet peak concentrations at the Split Lake outlet were as much as a factor of 3 above the aquatic life guideline. Although concentrations of some metals were higher at the Split Lake outlet than at upstream sites on the Nelson River, the extractable Mn concentration generally was lower, never approaching the drinking water guideline. These higher Al and Fe levels appear to be more characteristic of the Churchill and Burntwood River drainages than of the Nelson River (Section 3.5.3).

On the basis of the mean concentrations of total phosphorus, total nitrogen, and chlorophyll a, Split Lake can be classified as mesotrophic according to the model of Vollenweider (1979).

3.5.4.2 Water Chemistry - Water Discharge Relationships

3.5.4.2.1 Pre-development

Pre-development data for the outlet of Split Lake comes from 2 sites, one near the Split Lake community and the other near the outlet. Water quality - discharge relationships at these sites were examined separately.

Only four variables were correlated with discharge at the site near the Split Lake community, all negatively, in the 1972-1974 period. These included conductivity (lab), total alkalinity, total hardness, and dissolved K (Figure 39). The correlations accounted for 37 to 65% of the variance in the dependent variables.

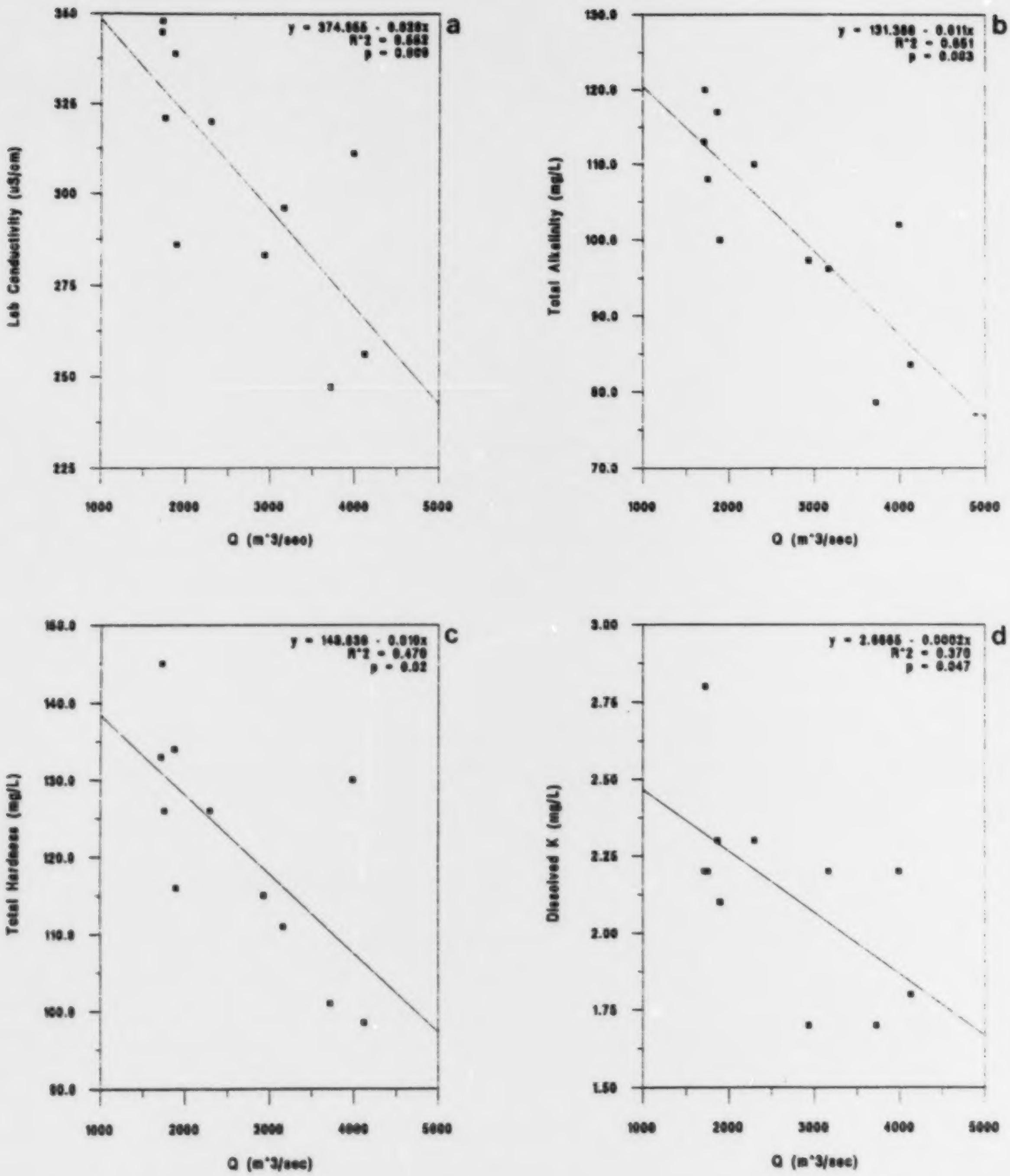


Figure 39. Water chemistry - water discharge relationships in Split Lake at Split Lake, 1972-1974.
a. lab conductivity b. total alkalinity c. total hardness d. dissolved K.

Six variables were negatively correlated with discharge at the Split Lake outlet in the 1972-1973 period (Figure 40). Two of these variables, conductivity (field) and dissolved K, were measured both here and at the site near the community and were correlated with discharge at both sites. Consistent with the correlation between hardness and discharge at the site near the community, Mg, a major component of hardness (APHA 1985), was correlated with discharge at the outlet. Other correlates with discharge at the outlet site included TDS and dissolved Na and SO_4. The regressions on discharge accounted for 66 to 92% of the variance in the dependent variables.

3.5.4.2.2 Current

Only 3 variables were significantly correlated with discharge in the 1987-1989 study period. All correlations were positive and included DOC, TOC, and dissolved Fe (Figure 41). The correlation between discharge and TOC was purely a result of the correlation with DOC. On average, 94% of TOC occurred as DOC (Table 27) resulting in significant collinearity of these variables. None of these relationships was particularly strong, as appears to be characteristic of sites influenced by the CRD, with the regression on discharge accounting 13 to 26% of the variance in the dependent variable.

The discharge relationship for dissolved Fe should not be used for predictive purposes, as it was clearly driven by the single high value (Figure 41c), unless the relationship can be verified by further sampling. The discharge relationships for TOC and DOC

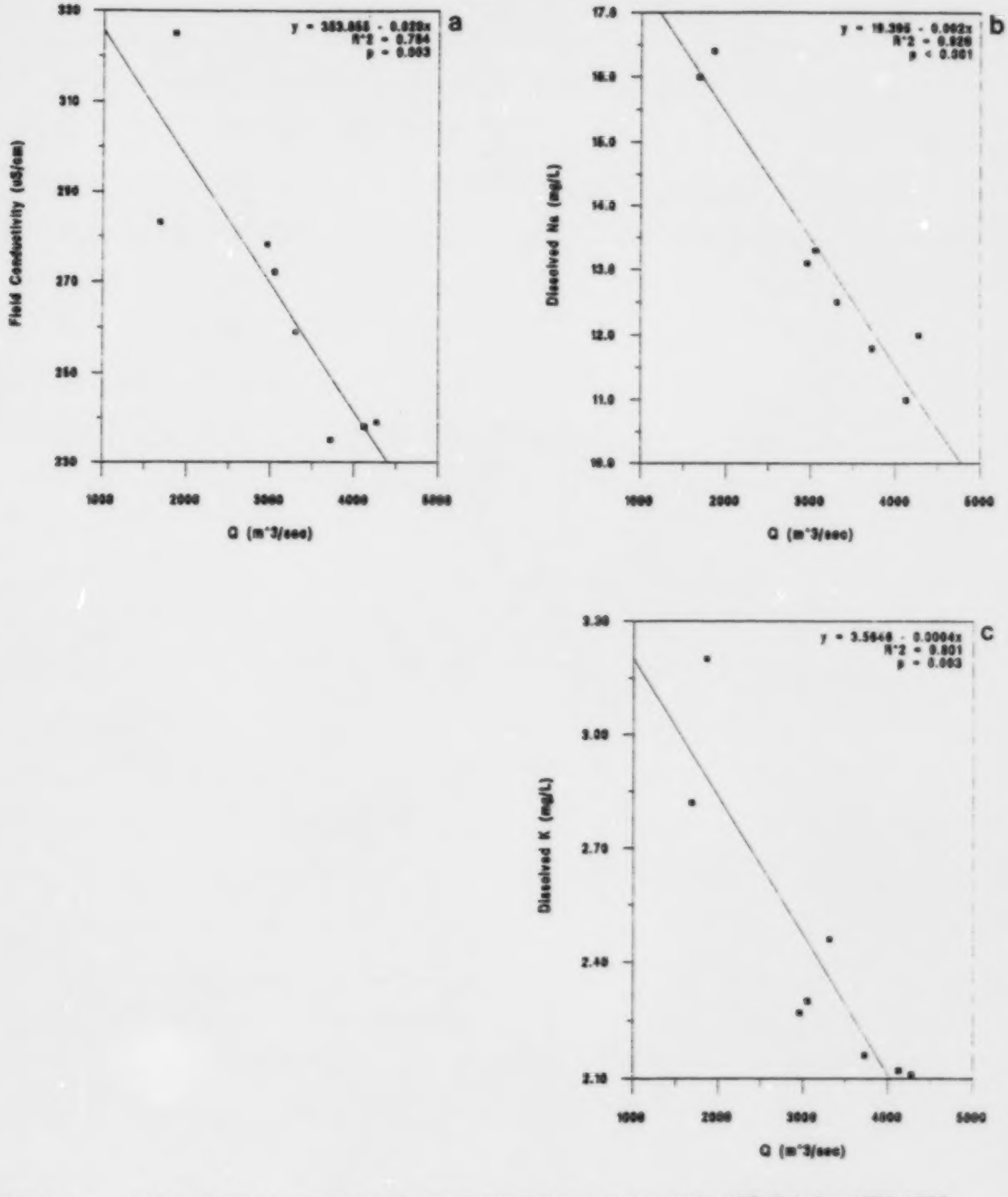


Figure 40. Water chemistry - water discharge relationships near the outlet of Split Lake, 1972-1973. a. field conductivity b. dissolved Na c. dissolved K d. dissolved Mg e. dissolved SO_4 f. TDS.

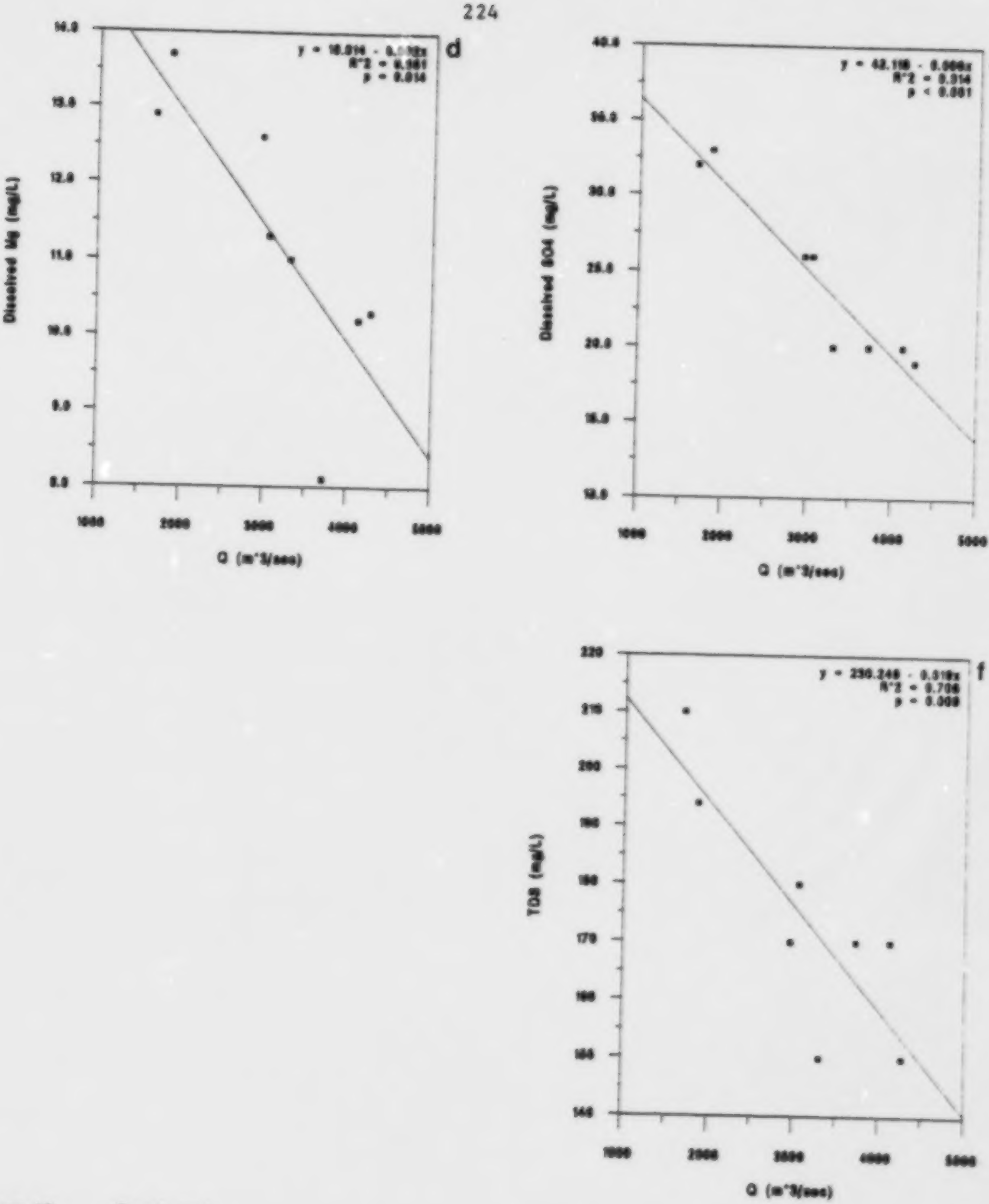


Figure 40. Continued.

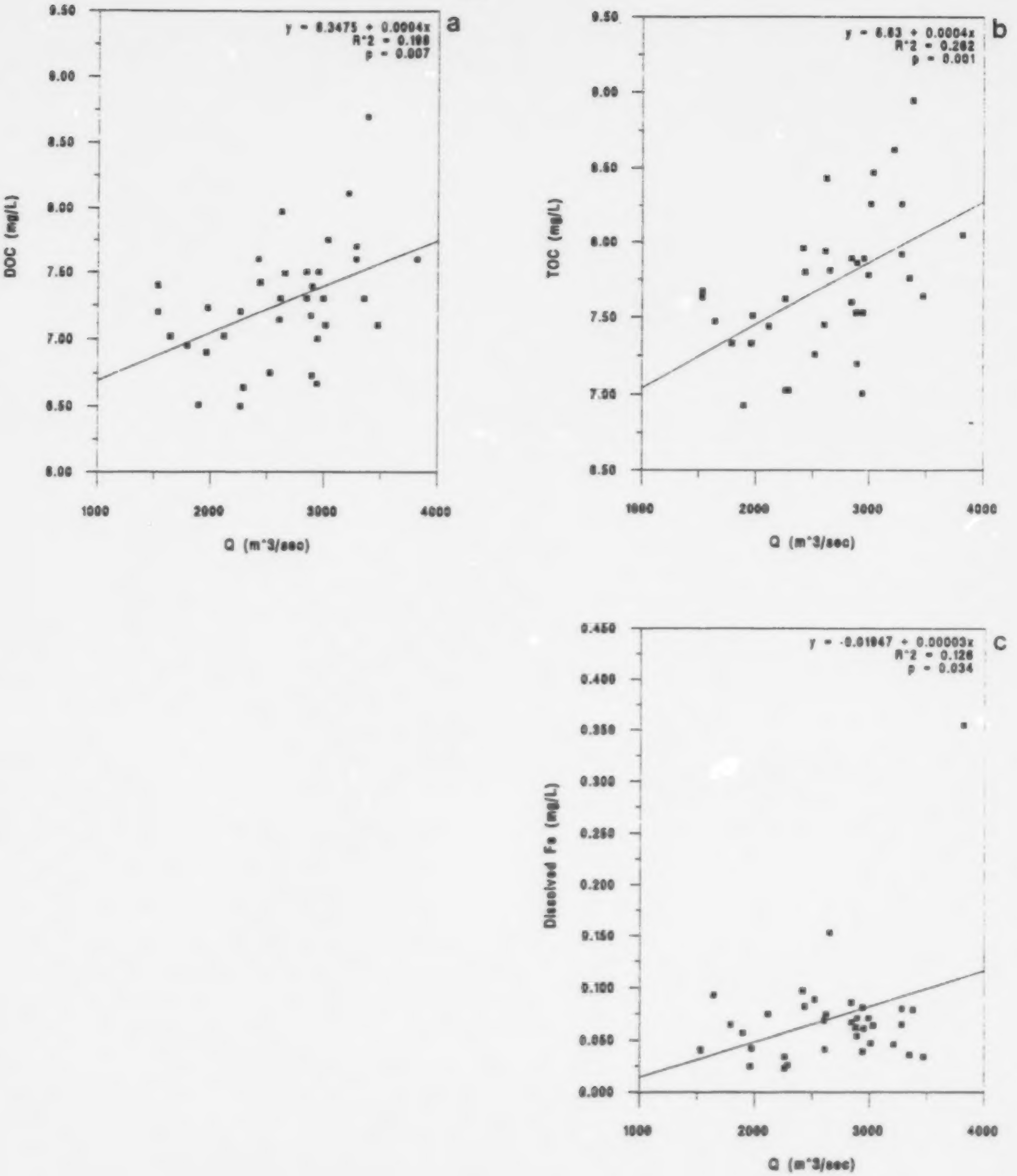


Figure 41. Water chemistry - water discharge relationships in the Nelson River at the outlet of Split Lake, 1987-1989. a. DOC b. TOC c. dissolved Fe.

appear to have some predictive ability as they are consistent with the general linear trends in the data.

Water chemistry - discharge relationships in the vicinity of the Split Lake outlet appear to have changed considerably between the 1972-1973 pre-development period and the 1987-1989 FEMP study. None of the variables which were correlated with discharge prior to development was correlated with discharge in the current study. All post-development relationships had a positive slope, compared to the negative slopes which prevailed before development. This shift in the nature of the discharge relationship, from negative to positive, occurred at other sites which have been affected by CRD. This change cannot conclusively be attributed to CRD, however, because other potential causes cannot be ruled out with the available data. Other potential causes include the low Nelson River flows during the FEMP study (Section 1.4), and the reversal of the hydrologic regime in the lake as a result of LWR. Discharge records for the pre-development period (1961-1975) indicate monthly mean discharge at the Kelsey Dam was 1852-2105 m^3 sec^{-1} under ice cover (Dec.- Mar.) compared with 2749-3202 m^3 sec^{-1} during the ice-free season (May-Oct.; Water Survey of Canada 1990). Since 1978, monthly mean flows through Split Lake (sum of Nelson R. and Burntwood R. flows) have been higher during the ice-cover season (2872-3089 m^3 sec^{-1}, Dec. to Mar.) than during the ice-free season (2402-2755 m^3 sec^{-1}, May to Oct., Water Survey of Canada 1990). It should be possible to separate the effects of low flows due to drought from those due to CRD and LWR when the drought conditions subside. In the meantime, the water quality - water discharge relationships based on the FEMP study should not be used for prediction or management purposes until the cause of the changes can be clarified.

3.5.4.3 Seasonal Differences

There appears to have been a change in the patterns of seasonal variation of several variables in Split Lake since CRD and LWR, although there is insufficient pre-development data on which to base a statistical analysis. Prior to development, conductivity, alkalinity (total), hardness (total), and dissolved Na, K, and SO_4 all were slightly higher under ice cover than during the open water season (Figure 42). These increases under ice cover appear to be the combined result of lower river flows in winter and the negative correlations of these variables with discharge.

In contrast to the pre-development period, few variables occurred in different concentrations under ice cover than in the open water season during the FEMP study, and none was common to the pre- and post-development periods. Those which showed some evidence of seasonal variation after development included TDN, dissolved NO_2-NO_3, and total N (which was comprised predominantly of dissolved species) and dissolved Zn, all of which occurred in higher concentrations under ice-cover (Figure 42). The occurrence of increased NO_2-NO_3 concentrations in winter, and consequently of TDN and total N, is generally consistent with the higher levels in both major inflows during the ice-cover period (Figures 33 and 38).

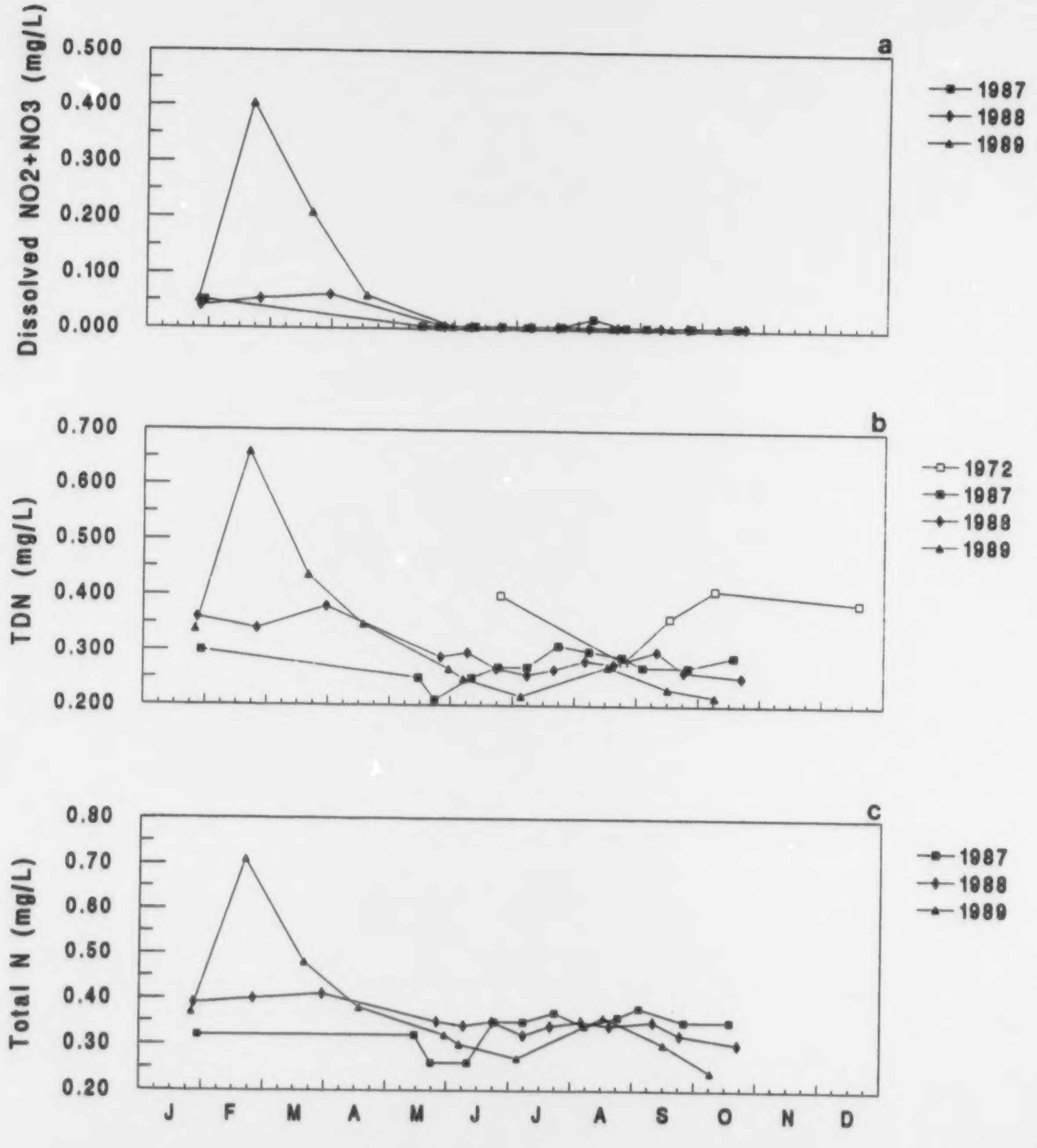


Figure 42. Seasonal water quality trends in the Nelson River at the outlet of Split Lake. a. dissolved NO_2-NO_3 b. TDN c. total N d. colour e. dissolved SO_4 f. dissolved Zn g. total hardness h. lab conductivity i. total alkalinity j. dissolved Ca k. dissolved K l. dissolved Na.

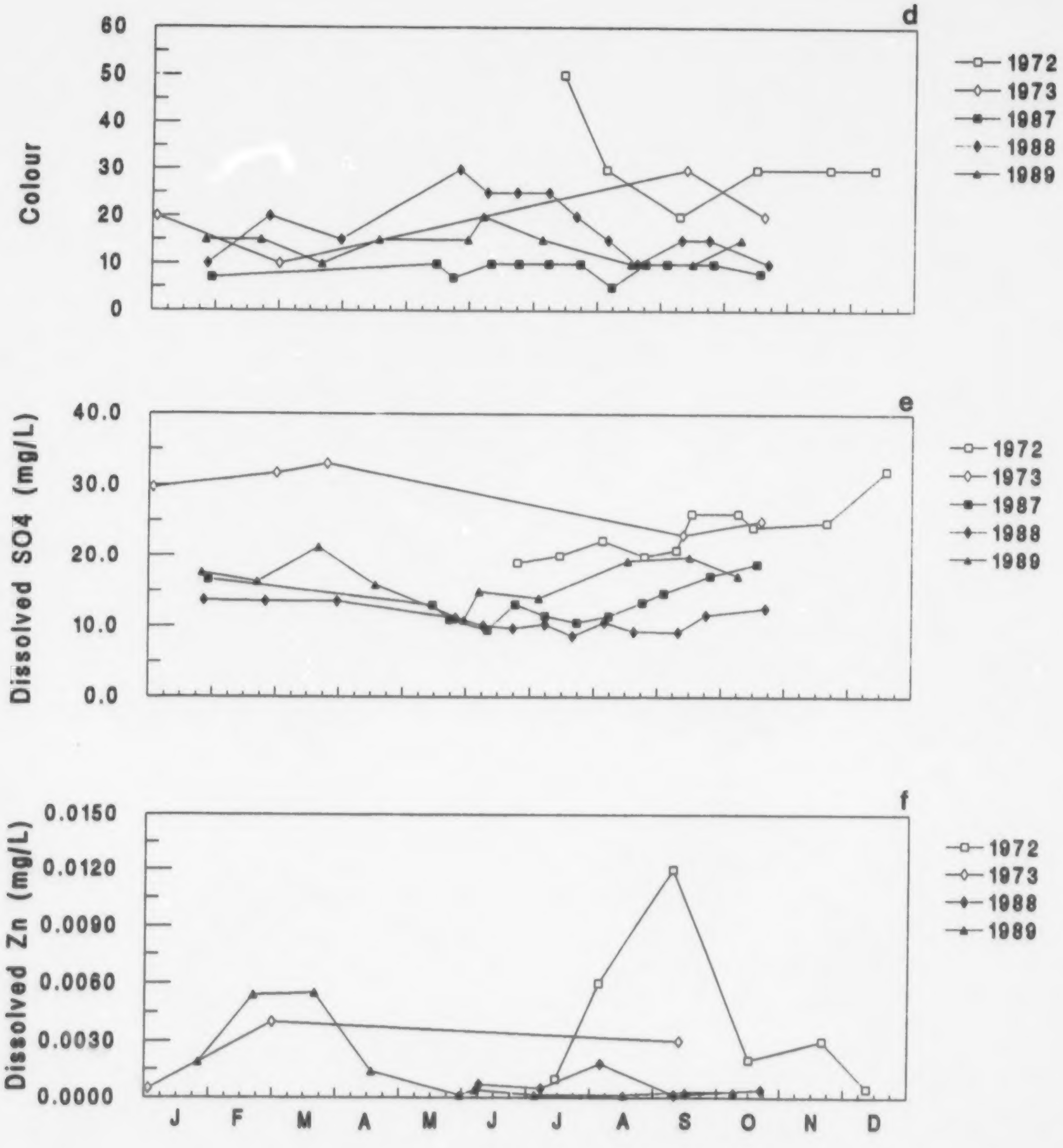


Figure 42. Continued.

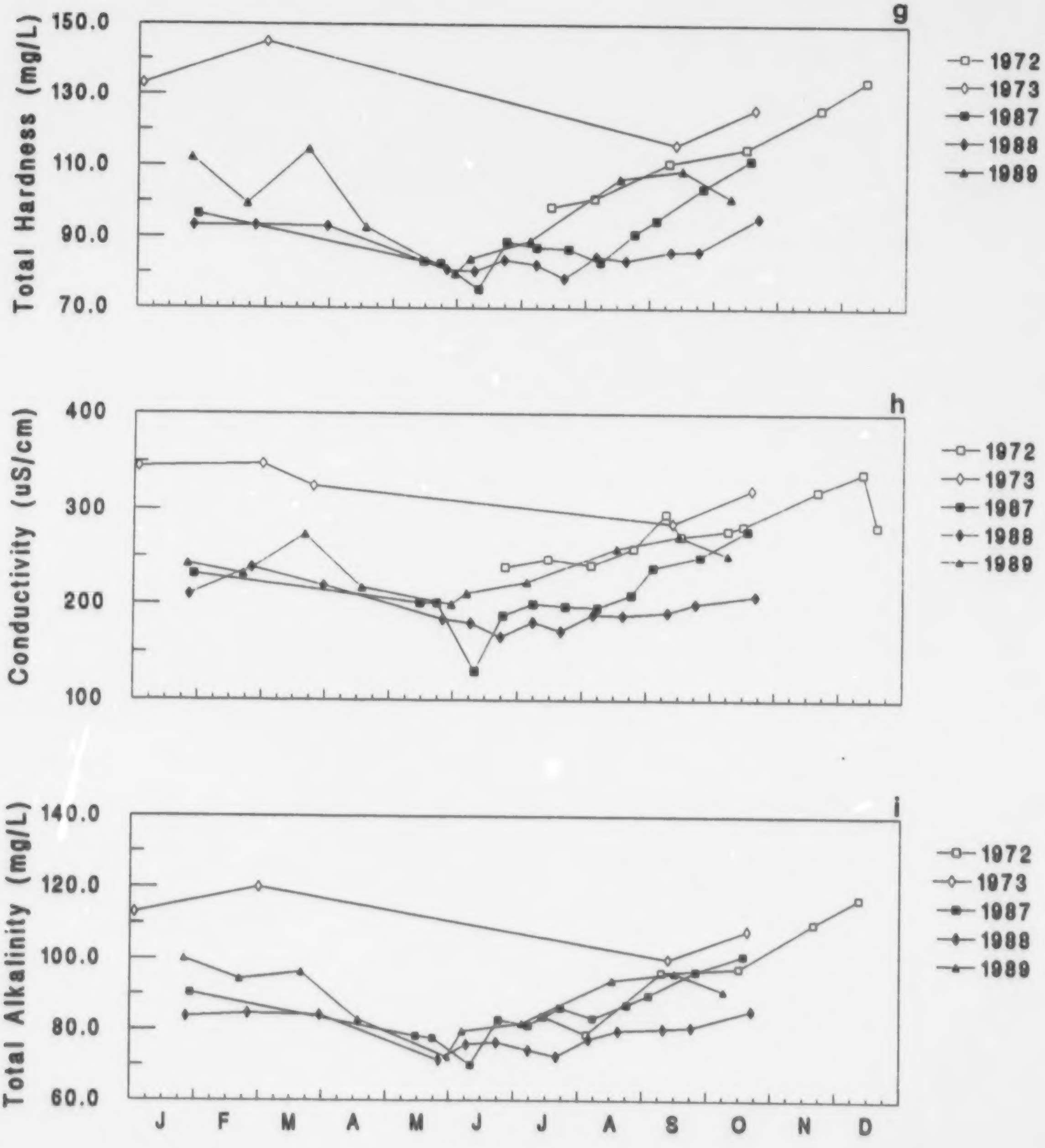


Figure 42. Continued.

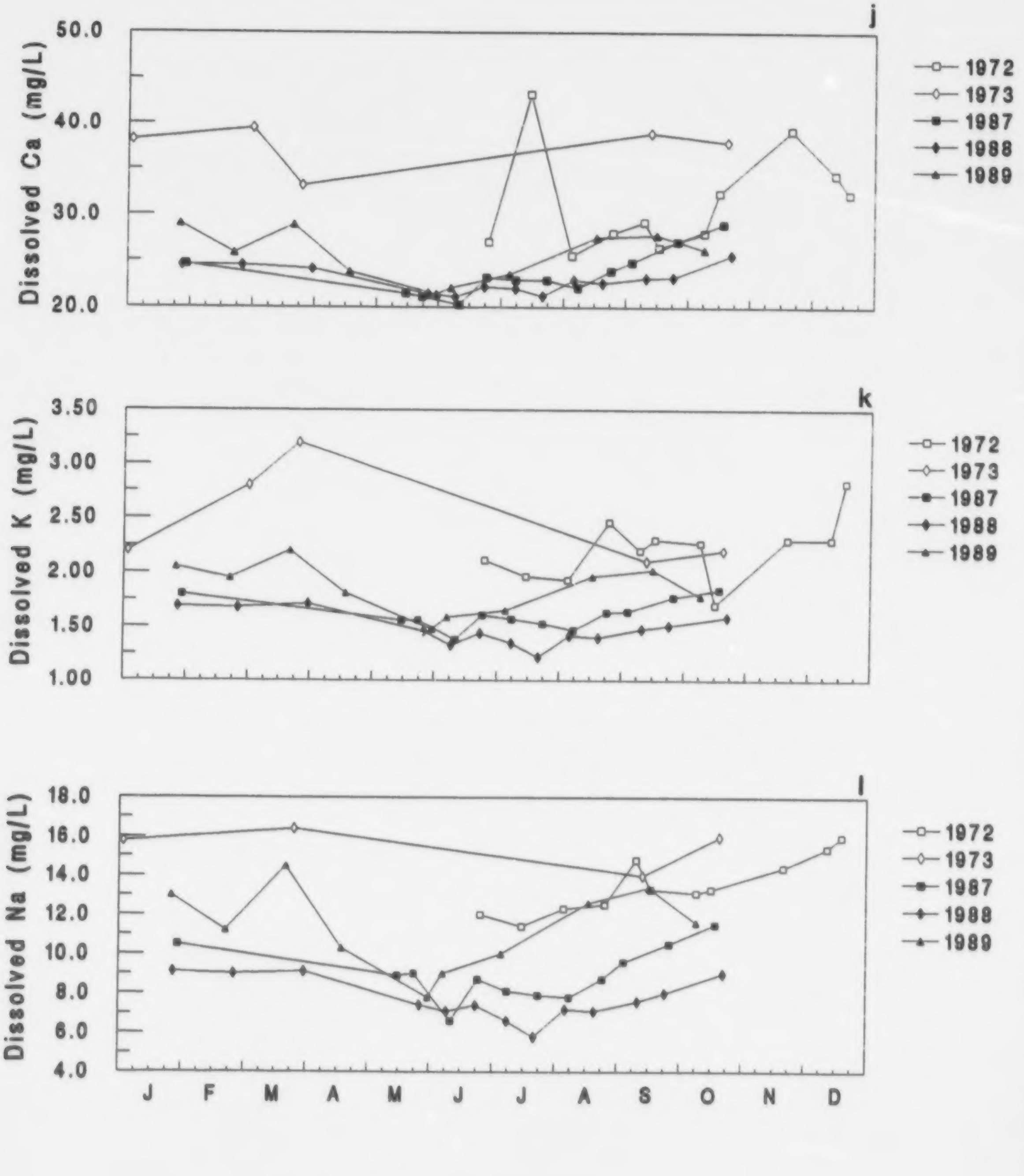


Figure 42. Continued.

Generation of NO_2-NO_3 within Split Lake also seems to play a role, as indicated by the much higher concentrations in February and March, 1989, than in the inflows. The winter maximum in dissolved Zn is not explained by higher loading, as no seasonal differences were evident in either major inflow at this time.

The absence of seasonal differences in several variables during the FEMP study could be due to one or more of the several changes in the water regime or the changes in water quality which have occurred since the pre-development study period. As noted in Section 1.4, Nelson River flows during the FEMP study period were the lowest in the 30 year record and this may have disrupted the normal discharge relationships and the accompanying seasonal trends. Another possible cause is the reversal of the natural seasonal flow regime due to operation of the LWR project. Peak flows now occur during the winter rather than during spring run-off and the annual range in monthly mean flow also has decreased since CRD and LWR came into operation. Prior to development, the difference between the highest and lowest monthly mean flows was 1350 m^3 sec^{-1}, but this difference dropped to 687 m^3 sec^{-1} after development (Water Survey of Canada 1990). These factors may have been sufficient to eliminate the rather small seasonal differences which prevailed prior to development. Finally, most of the changes in water quality caused by CRD involved the same variables in which seasonal trends were evident prior to development but not during the FEMP study, suggesting a common cause. Clearly, there is no way to identify the most probable cause of these changes in seasonality with the available data. This assessment will require additional sampling during periods of higher Nelson River flows to determine the role of the naturally low flows in these changes.

3.5.4.4 Changes in Water Quality Since Lake Winnipeg Regulation and Churchill River Diversion

3.5.4.4.1 Pre/Post Comparison

Significant changes in 10 water quality variables occurred in the outlet basin of Split Lake between the 1972-1973 pre-development studies and the 1987-1989 FEMP study. pH declined by 3% while conductivity, hardness (total), alkalinity (total), and dissolved Ca, Mg, K, Na, Cl, and SO_4 declined by 26, 24, 19, 29, 29, 27, 35, 38, and 47%, respectively (Table 28).

Berger and Ramsey (1991) also examined changes in water quality related to Churchill River diversion at 2 sites on Split Lake. Data for the June to September period collected by Manitoba Natural Resources, Fisheries Branch, in 1987, 1988, and 1989 were compared with pre-development data from 1972 and 1973 contained in the LWCNR study reports (Cleugh 1974; Hecky and Harper 1974; Morelli 1974), the same pre-development database employed in the present analysis. The sites examined were the outlet of Split Lake, near the sampling site for the FEMP study, and a site in the plume of the Burntwood River about 5 km from the inlet to Split Lake (Figure 5). Substantial reductions in major ion concentrations following CRD were found at the site near the Burntwood River inlet, with levels of Ca, Mg, Na, K, Cl, and SO_4 lower by 32, 48,

Table 28. Evaluation of the significance of changes in selected water quality parameters in the Nelson River near the outlet of Split Lake following Churchill River diversion and Lake Winnipeg regulation. X_{pre} denotes the mean for pre-development years, 1972 and 1973. X_{post} denotes the mean for the post-development years, 1987 through 1989. p is the probability of Student's t-statistic for the comparison.

Location		pH	Conductivity µS cm^{-1}	Turbidity NTU	Colour mg L^{-1}	NFR mg L^{-1}	NFRFixed mg L^{-1}	Tot Alk mg L^{-1}	Tot Hard mg L^{-1}	Diss Ca mg L^{-1}	Diss K mg L^{-1}	Diss Na mg L^{-1}	React Si mg L^{-1}
Nelson R.	p	0.008	0.001	0.794	0.136	0.183	0.563	0.104	<0.001	0.001	0.003	0.001	0.883
(Split L outlet)	X_{pre}	8.06	307	17	25	16	6	114.8	120.00	33.9	2.26	14.3	1.46
	X_{post}	7.81	228	18	14	10	8	84.3	91.7	24.1	1.65	9.3	1.51

		Diss Cl mg L^{-1}	Diss F mg L^{-1}	Diss SO$_4$ mg L^{-1}	Ortho P mg L^{-1}	Tot P mg L^{-1}	Diss NO$_2$NO$_3$ mg L^{-1}	TOC mg L^{-1}	Diss Cu mg L^{-1}	Diss Fe mg L^{-1}	Diss Pb mg L^{-1}	Diss Mn mg L^{-1}	Diss Zn mg L^{-1}
	p	0.020	0.076	0.002	0.239	0.206	0.079	0.077	0.534	0.829	0.053	0.275	0.721
	X_{pre}	15.3	0.08	26.0	0.004	0.103	0.345	9.58	0.010	0.072	0.004	0.015	0.004
	X_{post}	9.5	0.10	13.7	0.003	0.027	0.034	7.71	0.005	0.064	0.001	0.004	0.003

66, 46, 83, and 71%, respectively, due to increased Burntwood River flows. No other significant changes were detected at this site although a marginally significant reduction in NO_3-N and a marginal increase in TDP were indicated. Similar changes were found at the site near the Split Lake outlet, although the decreases were not as large as near the Burntwood River inlet due to additional mixing with the Nelson River. Levels of Na, Cl, SO_4, and colour were significantly lower, by 24, 36, 32, and 55%, respectively. It is not known if a similar decrease in colour occurred near the Burntwood River inlet due to the lack of pre-development data for this location. Mg was not included in the analysis of changes at the outlet station, again due to a shortage of pre-development data.

The lack of a significant change in Ca and K and the finding of a significant reduction in colour near the Split Lake outlet in the Berger and Ramsey (1991) study contrast with the significant reductions in Ca and K and lack of a significant change in colour in the present study. These discrepancies appear to be related to the restriction of the Berger and Ramsey (1991) study to the summer (June-September) season. Prior to CRD, Ca and K concentrations in Split Lake near the outlet tended to be higher in autumn and winter than in summer, with summer concentrations similar to those currently observed year-round (Figure 42). Thus, the reductions in the annual mean Ca and K levels detected in the present analysis were specifically due to lower levels during autumn and winter. In contrast, the reported reduction in colour appears to have been due to lower values in summer, but not necessarily in other seasons (Figure 42).

Berger and Ramsey (1991) also found water transparency at both sites, as indicated by Secchi disk visibility, was the same between 1987 and 1989 as in 1972-1973. Transparency was lower near the mouth of the Burntwood River than at other locations in the lake throughout the 1987-1989 study period, but this was the case prior to CRD as well. This absence of a change in transparency is consistent with the finding of no significant change in turbidity in the present study.

Similar changes to those found in this study and reported by Berger and Ramsey (1991) also were reported for several variables by Playle and Williamson (1986), with conductivity, hardness, alkalinity, and dissolved Ca, SO_4, and Mg declining by 19, 12, 27, 11, 36, and 18%, respectively. No significant change in colour was detected. In contrast to the present study, they did not find any significant change in the concentrations of dissolved K, Na, or Cl, and reported a significant 88% increase in turbidity.

The disparate conclusions of the Playle and Williamson (1986) study can be explained by a number of factors. Their different conclusion regarding a change in turbidity appears to have been due to the exclusion of some pre-development data. In particular, they did not include data for 1972 and 1973 reported in Morelli (1974). These data were not included in the analysis by Playle and Williamson (1986) because of a concern for a lack of data comparability indicated by different units of measurement. The data in Morelli (1974) were reported in Jackson Turbidity Units (JTU), compared with Nephthelometric Turbidity Units (NTU) in all later studies. Investigation revealed this concern was unfounded. Ed Sorba (Ward Lab, Manitoba Environment, pers. comm.) indicated the Morelli (1974) data were analyzed following the same procedure as later data. There have been changes in the type of nephthelometer used since 1974, but the method has otherwise remained unchanged. Although the Morelli (1974) turbidity values were reported in JTU, they were measured with a nephthelometer. On the basis of this information, the Morelli (1974) data were included in our analysis. When these data are added to the data considered by Playle and Williamson and reanalysed, the t-test also indicates no significant change ($p=0.773$; Figure 43).

All three studies concluded there has been no change in NFR at this site as a result of LWR and CRD. This appears consistent with the absence of significant change in turbidity or water transparency. Although NFR in the Burntwood River is significantly higher than in the Nelson River, the majority of this higher suspended sediment load appears to be lost through sedimentation shortly after entering Split Lake. Although all studies indicate that any impact of CRD on suspended sediment levels in Split Lake is not widespread, the zone of impact has not yet been adequately defined. Future studies should focus on the immediate area of delta formation.

Playle and Williamson (1986) did not detect a significant change in dissolved K because their data were the product of an

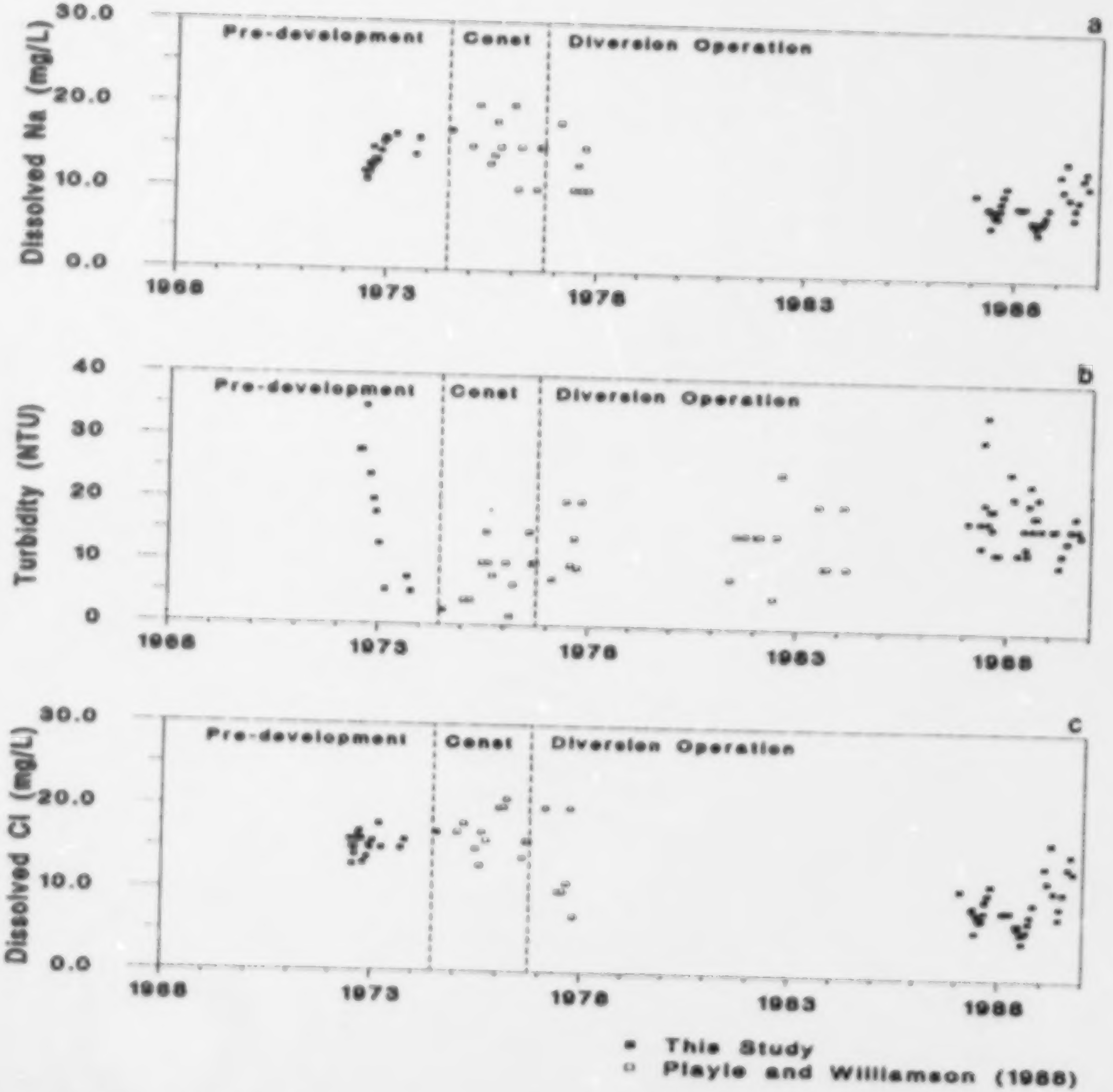


Figure 43. Temporal trends in (a) dissolved Na, (b) turbidity, and (c) dissolved Cl in the Nelson River at the outlet to Split Lake based on data analyzed by Playle and Williamson (1986) and the data analyzed in this study.

analytical method with inadequate sensitivity for the ambient K levels. The method had a detection limit of 2 mg L^{-1} (Playle 1986) which would not permit discrimination of the observed decrease from 2.26 to 1.65 mg L^{-1} (Table 28).

The lack of significant changes in Na and Cl in the Playle and Williamson (1986) study appears to have been a result of the short data records available to them. In particular, the post-development data records for these variables only extended from July, 1976, to October, 1977. Considering that full diversion flows were not achieved until September, 1977, their period of record was clearly too short to examine the effect of operational flows. In contrast, the present analysis is perhaps more indicative of the worst case impact of CRD on Na and Cl levels in Split Lake due to the unusually high relative contribution of the CRD to total Split Lake inflows during the study period.

Most of the changes in Split Lake water quality are solely attributable to the increased flows of softer water from the Churchill River drainage basin into the lake as a result of Churchill River diversion. Conductivity, hardness, and the concentrations of all major ions in the Burntwood River are significantly lower than in the Nelson River (Table 26), and no changes in conductivity or in dissolved Ca, Na, Cl, or SO_4 occurred in the Nelson River above Split Lake over the same period (Section 3.5.2.3). About 60% of the reduction in dissolved K near the outlet of Split Lake can be attributed to the lower concentration in the inflowing Nelson River, with the balance of the reduction due to the increased Burntwood River inflows. Only the small decrease in pH at the Split Lake outlet can be entirely attributed to changes in the Nelson River.

The decrease in colour during the summer is not directly attributable to CRD. Colour in the Burntwood River at the inlet to Split Lake was more than twice as high as in the Nelson River (Table 26). Insufficient pre-development colour data are available for the Nelson River to determine if there has been a reduction in this source.

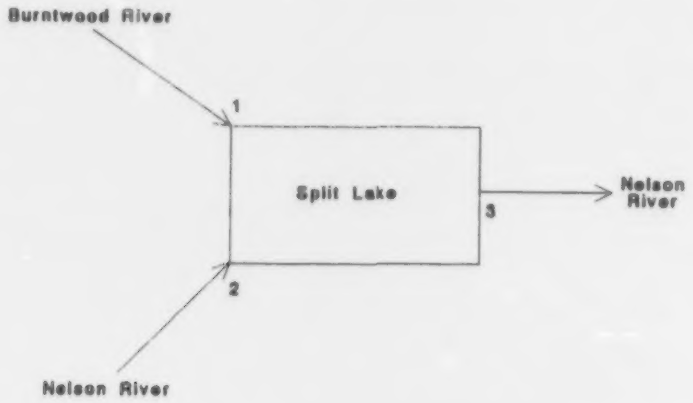
In summary, CRD has caused demonstrably significant reductions in hardness, alkalinity, and major ion concentrations in Split Lake through the introduction of softer Churchill River drainage water flowing to the lake. Although the quantity of sediment delivered to Split Lake via the Burntwood River has increased since CRD (Northwest Hydraulic Consultants 1988), this has not had a significant effect on turbidity, suspended sediment concentration (NFR), or water transparency in the majority of the lake due to sediment deposition in Split Lake near the mouth of the Burntwood River.

3.5.4.4.2 Modelled Changes

The dominance of Split Lake hydrology by the Nelson River prior to diversion, and now jointly by the Nelson and Burntwood rivers, makes it possible to predict changes in water quality which may have occurred near the outlet of Split Lake in the absence of pre-development data for this site. This modelling approach is based on the assumption that, prior to diversion, the concentration of a parameter at the outlet of Split Lake would have been approximately the same as in the Nelson River at the inlet to Split Lake. Therefore, the finding of a significant difference in the concentration of a parameter between the Nelson River inflow and the Split Lake outflow after diversion, in the absence of a pre/post comparison to the contrary, is evidence of a change due to CRD. This approach was judged to be reasonable on the basis of the dominance of total inflows to Split Lake by the Nelson River prior to diversion, the short residence time for the lake (approximately 6 days based on the lake volume, 1208.44 x 10^6 m^3 from Cherepak, 1990, and mean annual outflow volume from 1987 to 1989 of 71 km^3 $year^{-1}$, G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.) and the lack of a significant increase in shoreline erosion following diversion, as indicated by no change in turbidity, transparency, or NFR at the outlet.

For this approach to be valid, it must be shown that the present concentration of a variable in the Split Lake outflow is adequately predicted from the concentrations and discharges in the two major lake inflows, and that the pre-development concentration in the outflow was adequately predicted by the concentration in the Nelson River at the inflow to Split Lake. The present (post-diversion) concentration of a variable in the Split lake outflow was found to be readily predicted using the numerical model described in Figure 44. In this model, the only inflows considered are the Nelson and Burntwood rivers. Losses to sediments and

A. Simplified Model of Split Lake Hydrology



B. Model Derivation

1. Pre-development

a. Basic model, $Q_3C_3 = Q_2C_2 + Q_1C_1$
b. But, $Q_1 = 0$, relative to Q_2, which makes $Q_3C_3 = Q_2C_2$
c. Finally, $Q_3 = Q_2$, Therefore $C_3 = C_2$

2. Post-development

a. Basic model, $Q_3C_3 = Q_2C_2 + Q_1C_1$
b. Let $Q_1 + Q_2 = Q_3$, $Q_1 = 0.34 * Q_3$, and $Q_2 = 0.66 * Q_3$ - based on discharge data
c. Then $Q_3C_3 = (0.66\ Q_3)C_2 + (0.34Q_3)C_1$
d. Which reduces to $C_3 = 0.66C_2 + 0.34C_1$

Where Q_n = discharge at site 1, 2, or 3
C_n = concentration of substance (mg L^{-1}) at site 1, 2, or 3
Q_nC_n = flux of substance at site 1, 2, or 3

Figure 44. (A) Simplified model of Split Lake hydrology, and (B) derivation of numerical models used to predict the concentration of a substance in the Nelson River near the outflow of Split Lake based on the concentrations in the inflowing Burntwood and Nelson rivers and the relative contribution of these rivers to discharge at the outflow. Model predictions are compared with observed values in Table 29.

contributions from local inflows and direct precipitation were considered to be unimportant. The model was tested by comparing the predicted and observed annual mean concentrations in the Split Lake outflow for the 1987-1989 study period for all variables which had changed significantly between the pre- and post-diversion periods. The model predicted the concentration in the outflow with a mean (±1 s.d.) error of 5 (±3)% (Table 29), clearly indicating the importance of inflow volume and concentration to outflow chemistry.

The annual mean values for these variables at the Split Lake outlet prior to development were predicted equally well using a model which considered only the mean values in the Nelson River at the inlet to Split lake (Figure 44). The model predicted the mean value in the outflow with a mean (±1 s.d.) error of 7 (±5)% (Table 29). Therefore, the finding of a significant difference in concentration between the Nelson River inflow and the outlet of Split Lake during the 1987-1989 study period should be indicative of a change caused by diversion.

Among the many variables for which there are no pre-development data, only 3, dissolved As and extractable Fe and Mn, occurred in significantly different concentrations at the Split Lake outlet than at the Nelson River inlet during the 1987-1989 study period (Table 30). The excellent agreement between the observed and predicted outflow concentrations of dissolved As and extractable Fe (Table 29) indicates a direct effect of the Burntwood River on the concentrations of these variables in Split Lake. The increase in extractable Fe attributable to CRD was

Table 29. Comparison of observed and predicted values for selected water quality parameters in the Nelson River at the outlet of Split Lake following Churchill River diversion. Pre- and post-development predicted values have been calculated as in Figure 44. BWR denotes the Burntwood River. NR denotes the Nelson River. All values in mg L^{-1}, except where noted.

	Post-Development					Pre-Development		
	Inflow Value		Outflow Value			Outflow Value		
	BWR	NR	Predicted	Observed	% Error	Predicted	Observed	% Error
Conductivity ($\mu S\ cm^{-1}$)	123	293	235	228	3	273	307	10
TDS	65	155	124	119	4	177	171	4
Alkalinity (Total)	57	103	87	84	4		115	
Hardness (Total)	57	115	95	91	4		120	
DIC	13.4	24.4	20.7	19.9	4	22.2	22.7	2
Dissolved Ca	15.9	29.0	24.5	24.1	2	28.5	33.9	16
Dissolved Mg	4.1	10.2	8.1	7.7	5	11.1	10.9	2
Dissolved K	1.05	2.06	1.72	1.65	4	2.43	2.26	7
Dissolved Na	2.8	13.8	10.1	9.3	9	14.08	14.30	2
Dissolved Cl	1.2	15.3	10.5	9.5	11	17.23	15.3	13
Dissolved SO_4	2.94	21.1	14.9	13.7	9	24.5	26.0	6
Dissolved As	0.0002	0.0007	0.0005	0.0005	0			
Extractable Fe	0.787	0.193	0.395	0.387	2			
Extractable Mn	0.031	0.010	0.018	0.015	20			

Table 30. Comparison of current water quality in the Nelson River at the inlet and outlet to Split Lake. X_{in} denotes the 1987-1989 mean for the Nelson River at the inlet to Split Lake. X_{out} denotes the 1987-1989 mean for the Nelson River at the outlet to Split Lake. p is the probability of Student's t-statistic for the comparison.

Location		pH	Conductivity µS cm^{-1}	Turbidity NTU	Colour Relative	TDS mg L^{-1}	NFR mg L^{-1}	NFR Fixed mg L^{-1}	Tot Alk mg L^{-1}	Tot Hard mg L^{-1}	Diss Ca mg L^{-1}
Nelson R.	p	0.989	0.005	0.001	0.487	0.007	0.200	0.142	0.001	0.002	0.001
	X_{in}	7.85	293	9	9	154.7	5.0	3.3	103.0	115.0	29.3
	X_{out}	7.81	228	18	14	118.7	9.7	8.0	84.3	91.7	24.1
		Diss Mg mg L^{-1}	Diss K mg L^{-1}	Diss Na mg L^{-1}	React Si mg L^{-1}	Diss Cl mg L^{-1}	Diss F mg L^{-1}	Diss SO_4 mg L^{-1}	Ortho P mg L^{-1}	TDP mg L^{-1}	Tot P mg L^{-1}
	p	0.004	0.034	0.025	0.625	0.040	1.000	0.034	0.197	0.999	0.092
	X_{in}	10.2	2.06	13.8	1.24	15.3	0.10^{1}	21.1^{1}	0.004^{1}	0.011	0.022
	X_{out}	7.7	1.65	9.3	1.51	9.5	0.10	13.7	0.003	0.011	0.027
					TDN mg L^{-1}	Diss NO_2-NO_3 mg L^{-1}	Susp N mg L^{-1}	DOC mg L^{-1}	TOC mg L^{-1}	DIC mg L^{-1}	Chlorophyll a mg L^{-1}
	p				0.989	0.915	0.969	0.818	0.997	0.002	0.886
	X_{in}				0.305	0.016	0.048	7.51	7.79	24.4	0.002
	X_{out}				0.297	0.034	0.053	7.26^{1}	7.71^{1}	19.9	0.003

Table 30. continued.

Location		Extr Al mg L^{-1}	Diss Al mg L^{-1}	Diss As mg L^{-1}	Tot Cd mg L^{-1}	Diss Cd mg L^{-1}	Tot Co mg L^{-1}	Tot Cu mg L^{-1}	Diss Cu mg L^{-1}	Extr Fe mg L^{-1}	Diss Fe mg L^{-1}
Nelson R.	p	0.188	0.941	0.001	1.000	1.000	0.997	0.994	1.000	0.001	0.110
	X_{in}	0.186	0.055	0.0007	0.0001[1]	0.0001[1]	0.0003	0.0037	0.0020	0.193	0.039
	X_{out}	0.308	0.059	0.0005	0.0001	0.0001	0.0003	0.0051	0.0020	0.387	0.064[1]

	Tot Pb mg L^{-1}	Diss Pb mg L^{-1}	Extr Mn mg L^{-1}	Diss Mn mg L^{-1}	Tot Ni mg L^{-1}	Diss Ni mg L^{-1}	Diss Se mg L^{-1}	Tot V mg L^{-1}	Diss V mg L^{-1}	Tot Zn mg L^{-1}	Diss Zn mg L^{-1}
p	0.554	1.000	0.051	0.946	0.550	0.675	0.270	0.283	1.000	0.668	0.990
X_{in}	0.0006	0.0010	0.010	0.003	0.0011	0.0009	0.0002	0.0008	0.0005	0.0025	0.0010
X_{out}	0.0010	0.0010	0.015	0.004	0.0015	0.0011	0.0001	0.0011	0.0005	0.0041	0.0012

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[1] Variables standardized with discharge.

sufficient to cause the mean concentration at the outlet to exceed the guidelines for human consumption and protection of aquatic life by about 30%. Without CRD, the mean concentration at the outlet would have been about 35% below these guidelines given the present concentration in the inflowing Nelson River (Table 29). Based on the post-development model described in Figure 44, A 50% reduction in Burntwood River flows would be required for the mean extractable Fe concentration in Split Lake to comply with the guidelines.

Extractable Mn appears to behave less conservatively than Fe, with a 20% higher predicted concentration in the outlet than was observed (Table 29). Perhaps this 20% error was due to the sedimentation of Mn attached to particulates in the delta which is forming near the Burntwood River inlet. Although the impact is smaller than predicted, the higher concentration in the outflow than in the Nelson River is still attributable to the Burntwood River.

Sedimentation of particulate-bound metals would also explain the few significant differences in metals concentrations in the Nelson River between the inlet and the outlet to Split Lake (Table 30) despite the higher concentrations of several metals in the Burntwood River (Table 26). Site-specific investigations in the area of sediment deposition would be required to confirm this hypothesis.

3.5.5 Adverse Effects

The only adverse effect of Churchill-Nelson hydroelectric development on physical and chemical water quality in Split Lake has been the approximately two-fold increase in the extractable Fe concentration caused by CRD. This impact was most severe during the 1987-1989 sampling period, due to the high relative contribution of the CRD to Split Lake inflows. The elevation in extractable Fe is a product of the higher concentration in the Burntwood River and the increased contribution of this river to total Split Lake inflow due to diversion. As a result, the mean extractable Fe concentration in Split Lake exceeded the Canadian guidelines for drinking water and protection of aquatic life in the 1987-1989 study period. Although high extractable Fe is common in the northern drainages (i.e., Churchill, Burntwood, Footprint, and Aiken rivers; Sections 3.1 and 3.3), it is not characteristic of sites on the Nelson River.

It is unlikely that the elevated Fe concentration will have any toxic effects on aquatic life. The Canadian guideline (0.3 mg L^{-1}) is much more stringent than those of the Province of Manitoba (1.0 mg L^{-1}; Williamson 1988b) or the United States (1.0 mg L^{-1}; U.S. EPA 1976). The mean and maximum concentrations are well below the EC_{50} for fathead minnow eggs (1.5 mg L^{-1}), the most sensitive fish species tested which has been tested (CCREM 1987). Some invertebrates may be affected by concentrations as low as 0.32 mg L^{-1}, which is near the average for the 1987-1989 study period (CCREM 1987). However, the toxicity tests were conducted using dissolved

Fe in the absence of binding agents like suspended sediments. Most of the Fe in Split lake is in particulate form. The mean dissolved Fe concentration at the Split Lake outlet was just 0.07 mg L^{-1}, well below the level shown to have toxic effects on any species.

The effect of the increase in extractable Fe on the quality of drinking water for the Split Lake community is probably insignificant. The naturally high turbidity in the lake requires that the water be treated for turbidity removal prior to distribution. Surface water treatment plants operated for turbidity removal also remove Fe from raw waters (CCREM 1987). Most (>80%) of the Fe in Split Lake water is in particulate form, so turbidity removal should be effective in reducing Fe below the drinking water guideline.

The impact of the increased Fe concentration on those who use water directly from the lake is probably insignificant as well. Toxicity is not a significant concern, as the guideline level was set for aesthetic rather than health considerations (Health and Welfare Canada 1989). Potential impacts of excessive Fe include staining of laundry and plumbing, unpleasant taste and colour, and the promotion of undesirable biological growths within a waterworks (Health and Welfare Canada 1980). However, these impacts have not occurred upstream on the Burntwood River, where extractable Fe concentrations are naturally higher than in Split Lake. It should be possible to avoid any impact of the elevated Fe in Split Lake water by allowing particulates to settle out before use resulting in the removal of much of the Fe.

The perception among Split Lake community residents is that the lake has become more turbid since hydroelectric development on the Nelson and Churchill rivers began (Seidl et al. 1990). The appearance of a delta at the mouth of the Burntwood River since the diversion began operation is an obvious indication of the large quantities of suspended sediment entering the lake. However, this study and Berger and Ramsey (1991) have shown there has been no significant change in turbidity, water transparency, or NFR beyond the area of sediment deposition. The immediate area of the growing delta may be a site of severe adverse impact, but the nature and extent of any impacts in the delta area remain to be documented.

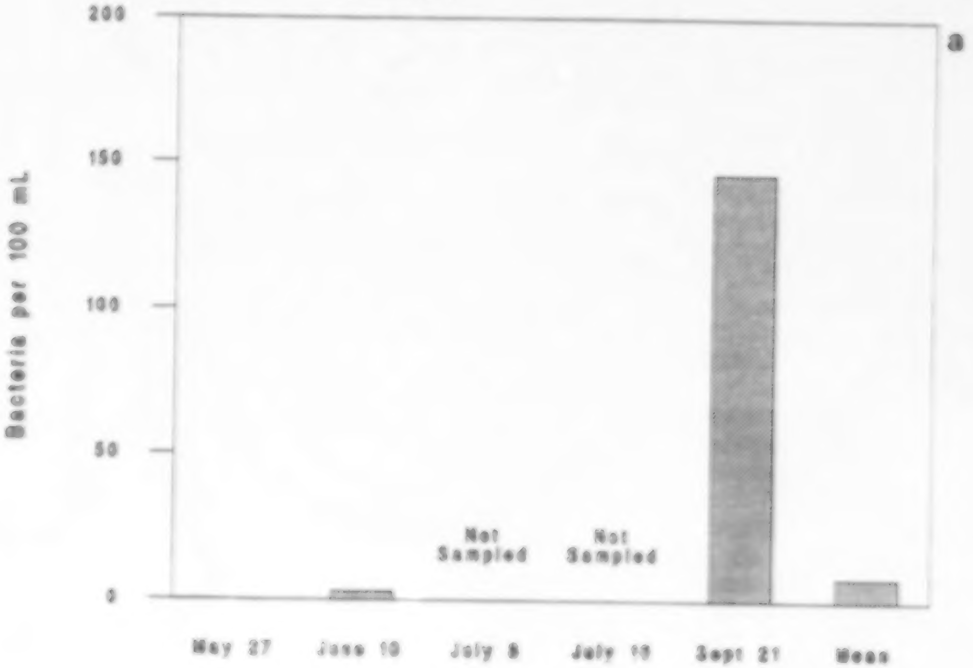
An aspect of the environmental effects of hydroelectric development in northern Manitoba which remains to examined is the effect of construction and operation of the Kelsey Generating Station on water quality and biota in Split Lake and other downstream waters. This study has shown CRD did not cause an elevation in levels of turbidity or NFR or a reduction in water transparency, yet the prevailing belief among Split Lake residents is that the water is more turbid now than in the past. It is possible that the perceived higher turbidity may have been caused by the Kelsey Dam rather than the diversion. The difficulty in examining this possibility is the virtual absence of data prior to construction of the dam, and the paucity of data for the immediate post-development period. Unless some other data can be found and validated, it may not be possible to completely document the effects of hydroelectric development on Split Lake.

None of the other significant changes in Split Lake water quality can be considered to represent an adverse effect. Indeed, the generally lower ionic content and hardness of Split Lake water since CRD could be interpreted as an improvement in water quality, albeit a relatively small one.

3.5.6 Split Lake Bacterial Testing

Geometric mean fecal coliform bacteria counts for the seasonal study ranged between 6 and 37 per 100 mL at the five sites (Figure 45). Counts at the Bay Store Beach Area, Beach Point (between nursing station and Bay store), and Second Beach (near ferry), were consistently below 200 per 100 mL. Counts in 4 of the 5 samples from Nursing Station Bay and First Beach also were well below 200 per 100 mL. Single high counts occurred at First Beach on May 27 (3,030), and at Nursing Station Bay on September 21 (217). Although water quality at the seasonal study sites appears to be excellent for recreational uses, as indicated by geometric mean counts below 200 bacteria per 100 mL, it is not possible to conclude that water quality is in strict compliance with the recreational water quality guideline because sampling frequency did not meet the guideline requirement of at least 5 samples within a 30 day period.

Intensive studies of bacterial water quality were conducted in the area of the sewage treatment plant outfall on August 12 and in waters adjacent to the proposed sewage lagoon on August 26, 1987. All fecal coliform counts were below 100 per 100 mL on August 12 with a geometric mean of 21 per 100 mL for the 10 samples (Figure



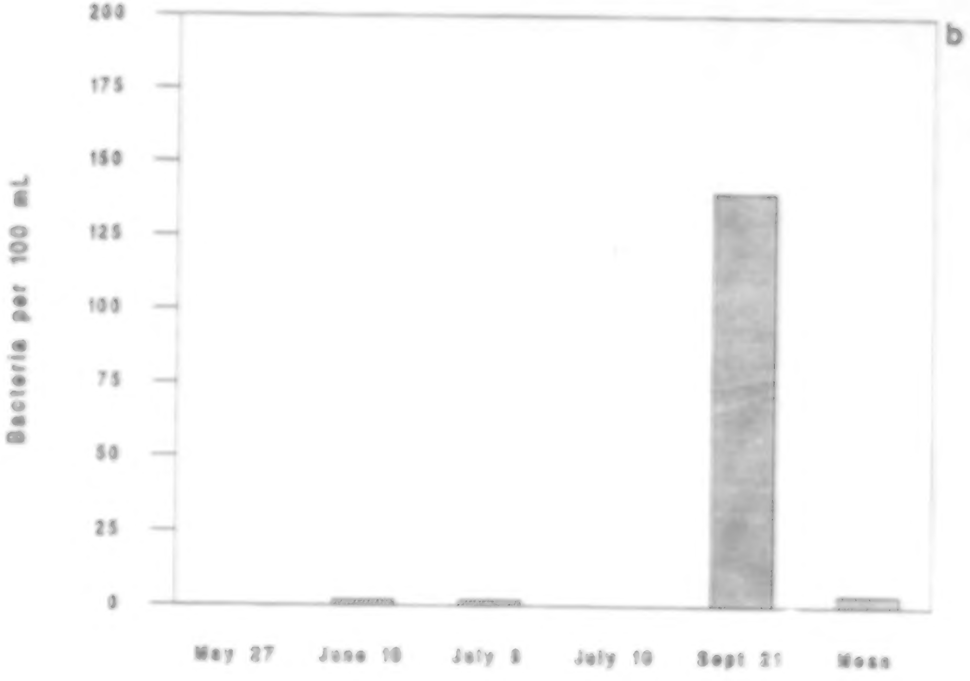
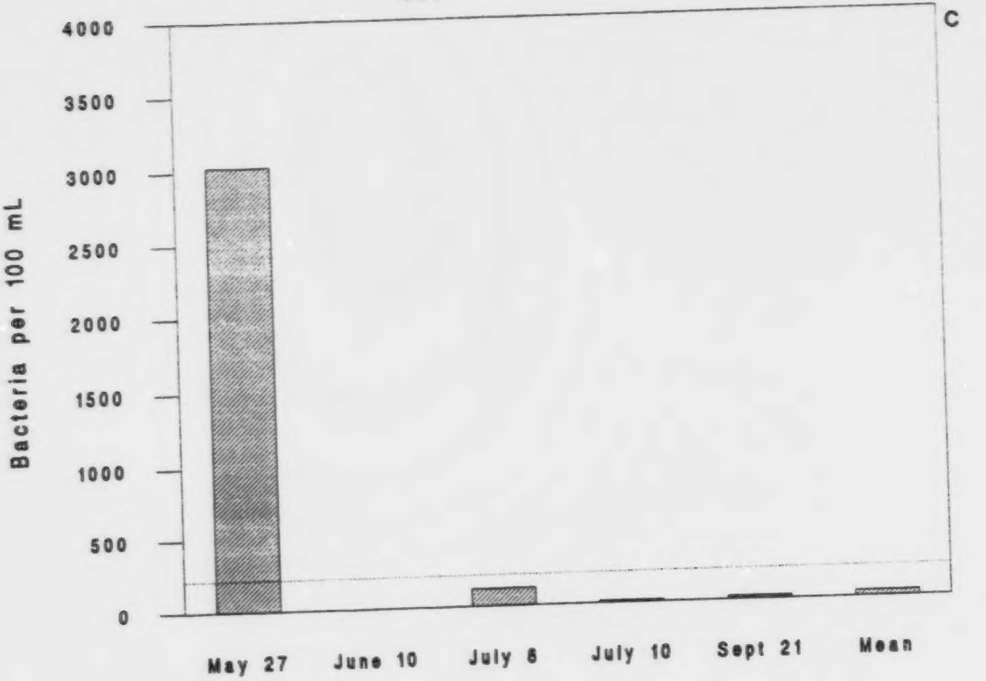


Figure 45. Fecal coliform bacteria counts from the seasonal study in the vicinity of Split Lake. Mean is the geometric mean. Dotted line indicates Canadian Water Quality Guideline. a. Bay Store Beach Area b. Beach Point between Nursing Station and Bay Store c. First Beach d. Nursing Station Bay e. Second Beach near Ferry. See Appendix 1 for site locations and descriptions.



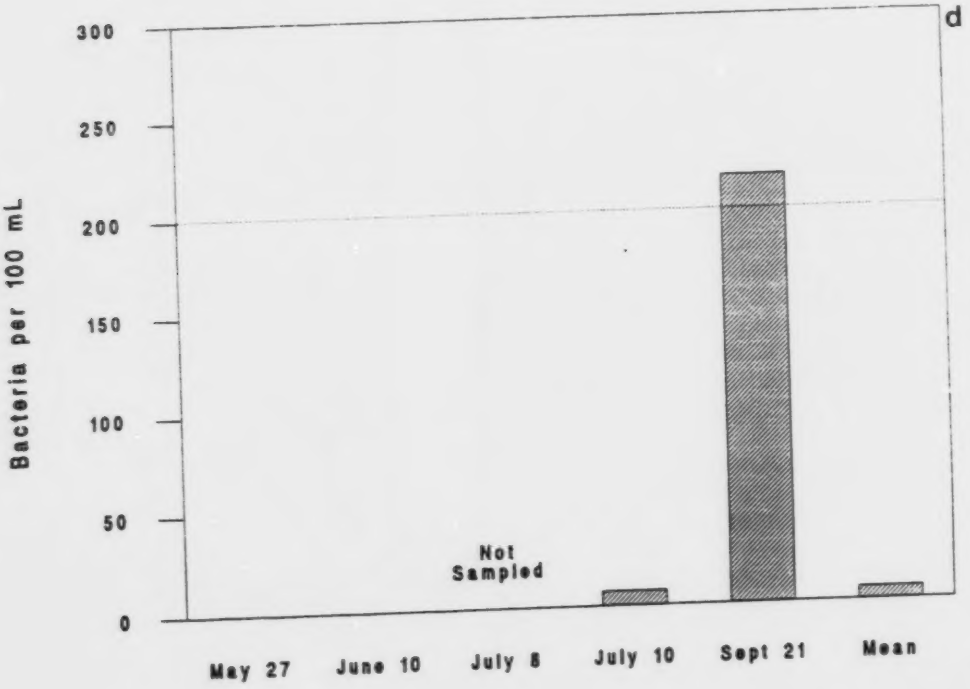


Figure 45. Continued.

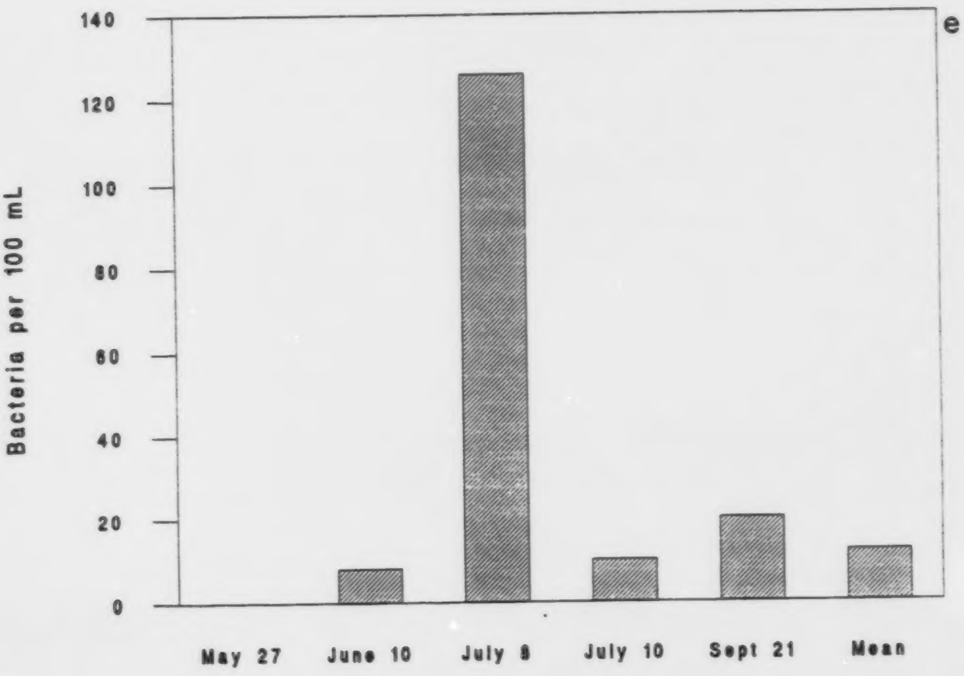


Figure 45. Continued.

46). Fecal coliform counts were below 200 bacteria per 100 mL in 9 of the 10 samples on August 26, with a very high count, 8480 bacteria per 100 mL, in one sample (Figure 46). The geometric mean count for the second survey was 8 bacteria per 100 mL. Bacteriological water quality at these sites appears to be excellent for recreational uses with geometric mean fecal coliform counts at 5-10% of the Canadian guideline.

3.5.7 York Landing Bacterial Testing

Geometric mean fecal coliform bacteria counts for the study season ranged between 1 and 10 per 100 mL at the survey sites at York Landing (Figure 47). Counts were consistently below 20 per 100 mL in all samples from the Gravel Pit, Water Treatment Plant, and Island Beach, and in 4 of the 5 samples from the Ferry Dock. The single high value at Ferry Dock, 460 per 100 mL, occurred on September 21. No intensive surveys were conducted at York Landing. The very low geometric mean fecal coliform counts in comparison to the Canadian recreational water quality guideline of 200 bacteria per 100 mL suggests bacteriological water quality is excellent for recreational uses. However, as was the case at the other 4 communities, sampling was less frequent than the 5 samples per 30 day period specified by the guideline so it is not known if water quality was in strict compliance. Any future studies should ensure that sampling meets the guideline requirements.

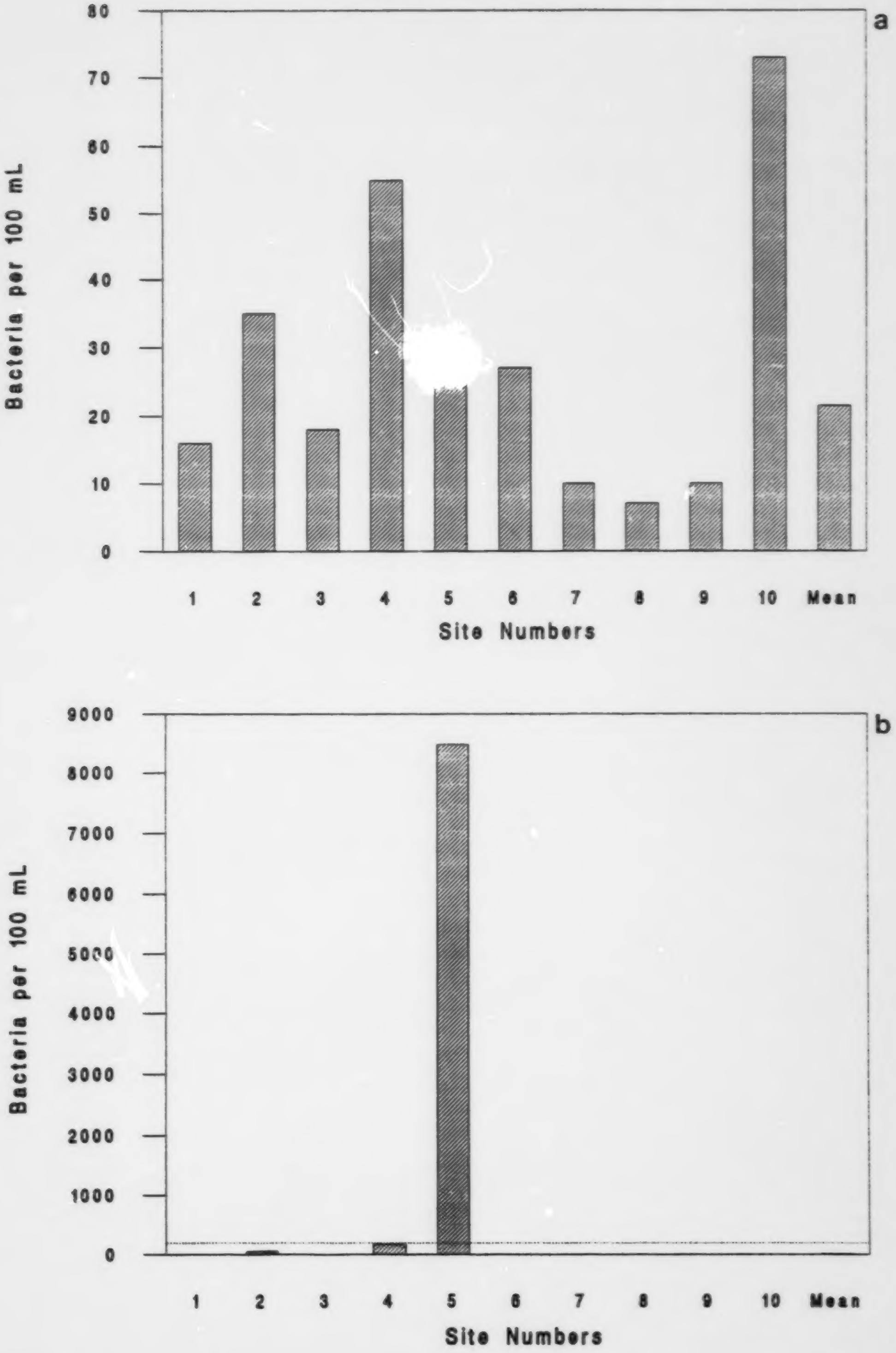


Figure 46. Fecal coliform bacteria counts from the intensive studies in the vicinity of the Split Lake Sewage Treatment Plant Outfall. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Sampled 12 August 1987 b. Sampled 26 August 1987. See Appendix 1 for site locations and descriptions.

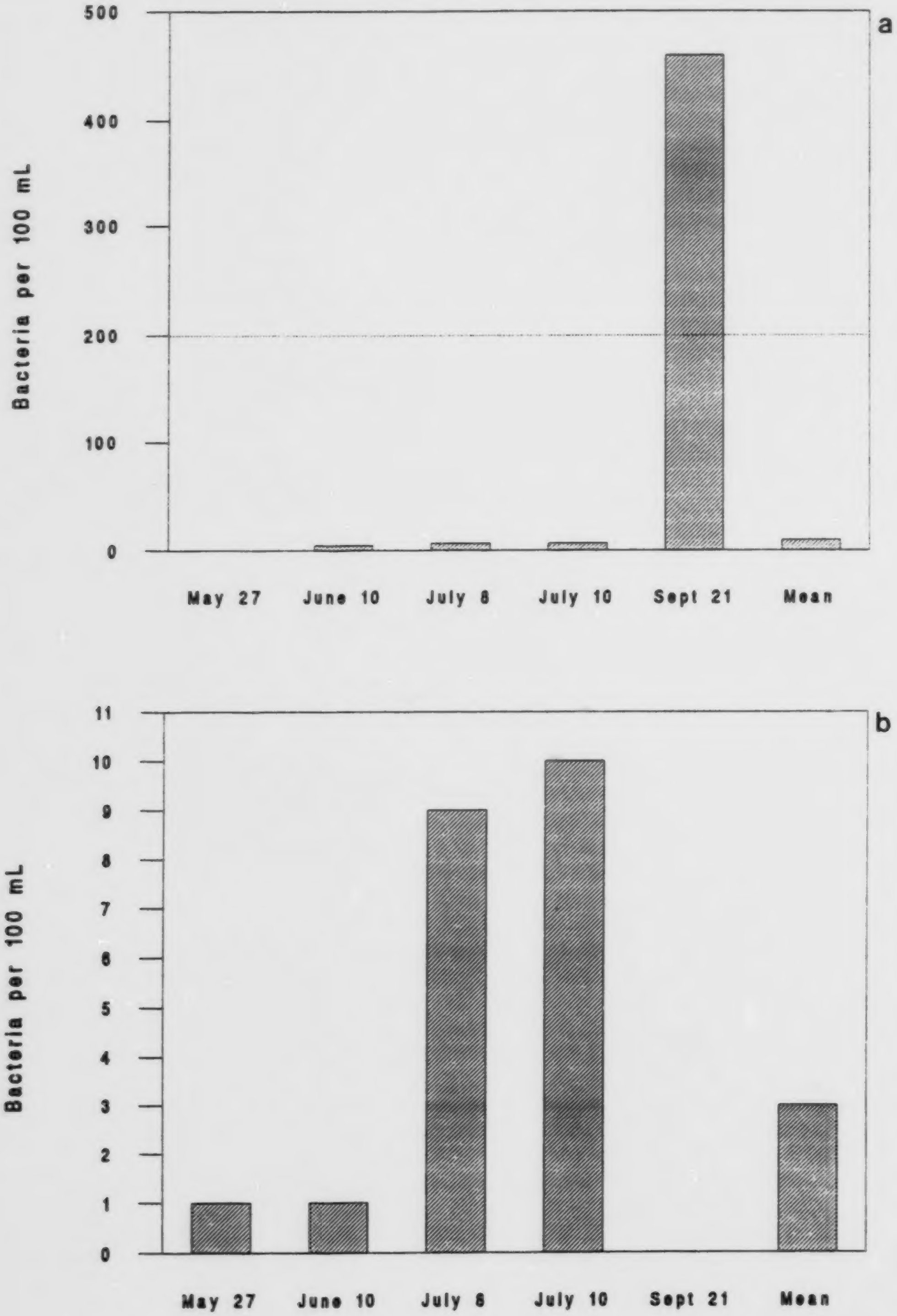


Figure 47. Fecal coliform bacteria counts from the seasonal study in the vicinity of York Landing. Mean is the geometric mean. Dotted line indicates the Canadian Water Quality Guideline. a. Ferry Dock b. Gravel Pit c. Island Beach d. Water Treatment Plant. See Appendix 1 for site locations and descriptions.

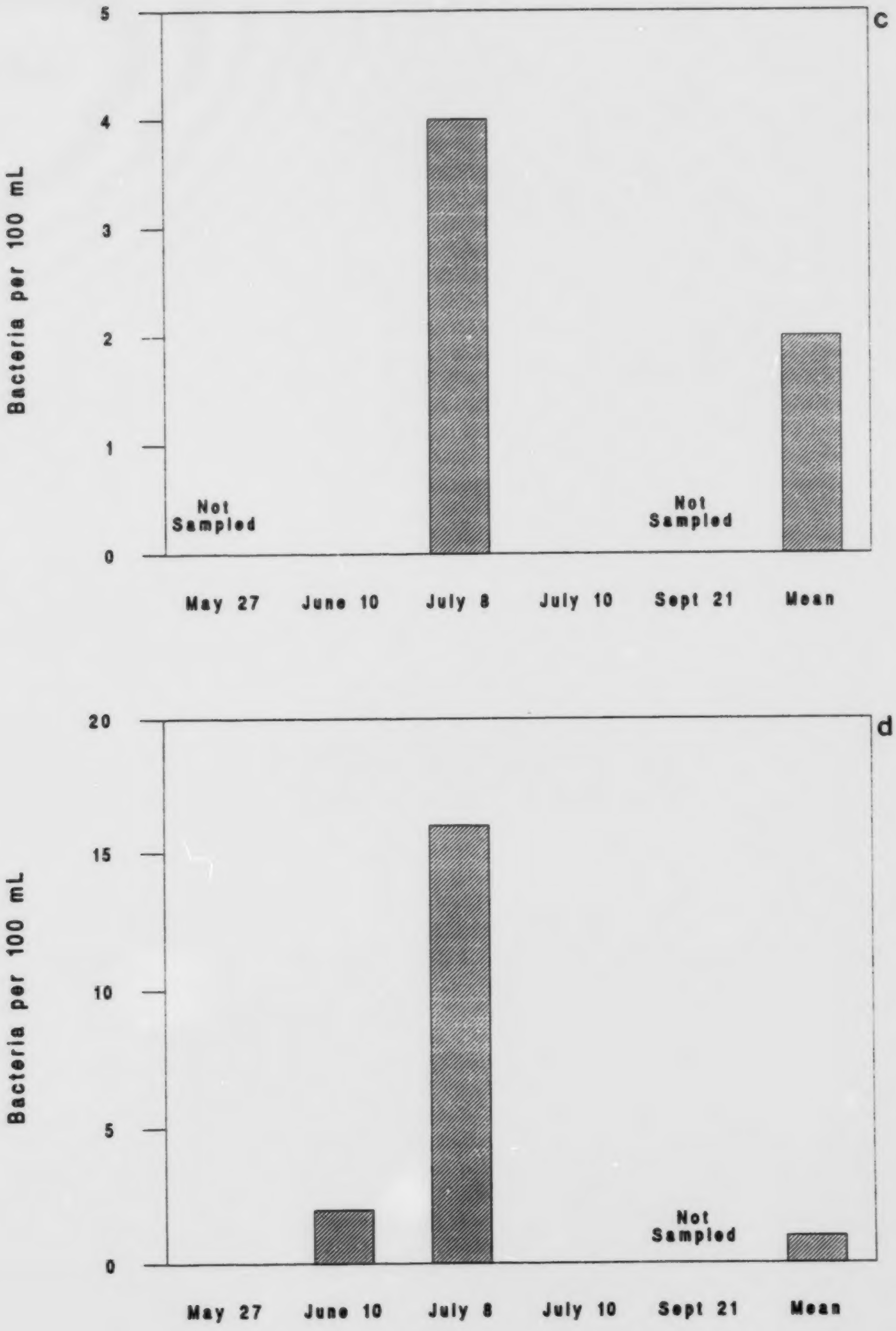


Figure 47. Continued.

4.0 Conclusions

4.1 Current Water Quality

Physical and chemical water quality at all 11 sites generally meets the Canadian Water Quality Guidelines for drinking water, recreational uses, and protection of aquatic life although there are a number of exceptions. Mean turbidity exceeded the Canadian drinking water guideline and mean extractable Al and total Cu exceeded the aquatic life guidelines at all sites (Tables 31, 32, 33, and 34). At the natural river sites and on the upper Nelson River (i.e., above Split Lake), mean turbidity was in the range of 6 to 19 NTU. Turbidity on the CRD between Threepoint and Split lakes was higher, ranging from 20 to 35 NTU, due to shoreline and bank erosion caused by the diversion. Extractable Al and total Cu concentrations at sites where the hydrologic regime was altered by hydroelectric development generally were within the ranges found in the natural rivers.

The natural rivers and sites on the CRD were distinguished by mean extractable Fe exceeding the drinking water and aquatic life guidelines. Mean extractable Fe was below the drinking water and aquatic life guidelines in the Nelson River at sites above Split Lake and in the Jack River. Both guidelines were exceeded in the Nelson River at the outlet of Split Lake. The high levels in Split Lake appear to be a direct result of CRD. Similarly, sporadic exceedences of the aquatic life guidelines for total Pb, Cd, and Zn occurred in the natural rivers and at sites on the CRD, but never in the Jack River or in the Nelson River above Split Lake. Total Pb

Table 31. Water quality variables in the natural rivers which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variables	Units	Mean	s.d.	Maximum	Minimum
Aiken River at the inlet to Split Lake					
Turbidity (Lab)	NTU	6.0^{+}	4.0	18.0^{+}	1.0^{+}
Extractable Al	mg L^{-1}	0.107^{+++}	0.059	0.226^{+++}	<0.100
Total Cu	mg L^{-1}	0.0029^{+++}	0.0041	0.0170^{+++}	<0.0005
Extractable Fe	mg L^{-1}	0.438$^{+,+++}$	1.008	5.980$^{+,+++}$	0.101
Extractable Mn	mg L^{-1}	0.093^{+}	0.339	1.920^{+}	0.005
Burntwood River above Threepoint Lake					
Turbidity (Lab)	NTU	19.0^{+}	4.0	27.0^{+}	13.0^{+}
Extractable Al	mg L^{-1}	0.443^{+++}	0.148	0.700^{+++}	0.191^{+++}
Total Cd	mg L^{-1}	0.0001	0.0002	0.0009^{+++}	<0.0001
Total Cu	mg L^{-1}	0.0061^{+++}	0.0081	0.0296^{+++}	0.0012
Extractable Fe	mg L^{-1}	0.642$^{+,+++}$	0.165	1.050$^{+,+++}$	0.183
Total Pb	mg L^{-1}	0.0009	0.0007	0.0028^{+++}	<0.0007
Extractable Mn	mg L^{-1}	0.041	0.016	0.090^{+}	0.005
Footprint River above Footprint Lake					
Turbidity (Lab)	NTU	9.0^{+}	3.0	18.0^{+}	4.0
Extractable Al	mg L^{-1}	0.190^{+++}	0.097	0.410^{+++}	<0.100
Total Cu	mg L^{-1}	0.0232^{+++}	0.1060	0.5530^{+++}	0.0005
Extractable Fe	mg L^{-1}	0.420$^{+,+++}$	0.246	1.150$^{+,+++}$	0.091
Total Pb	mg L^{-1}	0.0011	0.0030	0.0161^{+++}	<0.0007
Extractable Mn	mg L^{-1}	0.057^{+}	0.078	0.296^{+}	<0.002
Total Zn	mg L^{-1}	0.0040	0.0099	0.0515^{+++}	0.0004

$^{+}$ Exceeds limit for drinking water.

$^{++}$ Exceeds limit for recreational use.

$^{+++}$ Exceeds limit for protection of aquatic life.

Table 32. Water quality variables at sites in the Norway House area which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variable	Units	Mean	s.d.	Maximum	Minimum
Nelson River below Sea River Falls					
Turbidity (Lab)	NTU	10.0^{+}	3.0	16.0^{+}	6.0^{+}
Extractable Al	mg L^{-1}	0.181^{+++}	0.094	0.385^{+++}	<0.100
Total Cu	mg L^{-1}	0.0047^{+++}	0.0054	0.0208^{+++}	0.0011
Extractable Fe	mg L^{-1}	0.197	0.072	$0.365^{+,+++}$	0.087
Extractable Mn	mg L^{-1}	0.016	0.012	0.067^{+}	0.005
Jack River (West Channel)					
Turbidity (Lab)	NTU	9.0^{+}	5.0	26.0^{+}	3.0
Extractable Al	mg L^{-1}	0.176^{+++}	0.126	0.442^{+++}	<0.100
Total Cu	mg L^{-1}	0.0034^{+++}	0.0038	0.0159^{+++}	0.0011
Extractable Fe	mg L^{-1}	0.187	0.106	$0.457^{+,+++}$	0.070
Total Pb	mg L^{-1}	0.0007	0.0006	0.0027^{+++}	<0.0007

$^{+}$ Exceeds limit for drinking water.
$^{++}$ Exceeds limit for recreational use.
$^{+++}$ Exceeds limit for protection of aquatic life.

Table 33. Water quality variables at sites on the Churchill River Diversion which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variable	Units	Mean	s.d.	Maximum	Minimum
Rat River above Threepoint Lake					
Turbidity (Lab)	NTU	17.0^{+}	8.0	46.0^{+}	11.0^{+}
Extractable Al	mg L^{-1}	0.357^{+++}	0.116	0.712^{+++}	0.161^{+++}
Total Cd	mg L^{-1}	<0.0001	0.0001	0.0004^{+++}	<0.0001
Total Cu	mg L^{-1}	0.0053^{+++}	0.0091	0.0400^{+++}	0.0005
Extractable Fe	mg L^{-1}	$0.435^{+,+++}$	0.131	$0.926^{+,+++}$	0.143
Total Pb	mg L^{-1}	0.0008	0.0006	0.0021^{+++}	<0.0007
Extractable Mn	mg L^{-1}	0.019	0.013	0.081^{+}	0.004
Burntwood River at the outlet to Threepoint Lake					
Turbidity (Lab)	NTU	23.0^{+}	25.0	$160.0^{+,++}$	10.0^{+}
Extractable Al	mg L^{-1}	0.377^{+++}	0.209	1.360^{+++}	<0.100
Total Cd	mg L^{-1}	0.0001	0.0001	0.0006^{+++}	<0.0001
Total Cu	mg L^{-1}	0.0044^{+++}	0.0059	0.0264^{+++}	0.0005
Extractable Fe	mg L^{-1}	$0.477^{+,+++}$	0.303	$2.010^{+,+++}$	0.135
Total Pb	mg L^{-1}	0.0008	0.0009	0.0046^{+++}	<0.0007
Extractable Mn	mg L^{-1}	0.020	0.028	0.179^{+}	0.006
Burntwood River at Thompson					
Turbidity (Lab)	NTU	27.0^{+}	6.0	37.0^{+}	15.0^{+}
Extractable Al	mg L^{-1}	0.491^{+++}	0.154	0.826^{+++}	0.230^{+++}
Total Cu	mg L^{-1}	0.0040^{+++}	0.0051	0.0240^{+++}	0.0009
Extractable Fe	mg L^{-1}	$0.624^{+,+++}$	0.189	$0.931^{+,+++}$	0.256
Total Pb	mg L^{-1}	0.0009	0.0006	0.0027^{+++}	<0.0007

$^{+}$ Exceeds limit for drinking water.
$^{++}$ Exceeds limit for recreational use.
$^{+++}$ Exceeds limit for protection of aquatic life.

Table 34. Water quality variables at sites in the Split Lake area which exceeded one or more of the Canadian Water Quality Guidelines. Listed values are the mean, standard deviation (s.d.), maximum and minimum values during the 1987-1989 FEMP study period.

Variable	Units	Mean	s.d.	Maximum	Minimum
Burntwood River at the inlet to Split Lake					
Turbidity (Lab)	NTU	35.0^{+}	10.0	$71.0^{+,++}$	20.0^{+}
Extractable Al	mg L^{-1}	0.582^{+++}	0.315	2.140^{+++}	0.235^{+++}
Total Cd	mg L^{-1}	<0.0001	0.0001	0.0003^{+++}	<0.0001
Total Cu	mg L^{-1}	0.0048^{+++}	0.0062	0.0278^{+++}	0.0010
Extractable Fe	mg L^{-1}	$0.774^{+,+++}$	0.416	$2.860^{+,+++}$	0.299
Total Pb	mg L^{-1}	0.0010	0.0007	0.0034^{+++}	<0.0007
Extractable Mn	mg L^{-1}	0.030	0.027	0.177^{+}	0.004
Nelson River below Grass River					
Turbidity (Lab)	NTU	9.0^{+}	3.0	10.0^{+}	5.0
Extractable Al	mg L^{-1}	0.188^{+++}	0.101	0.418^{+++}	<0.100
Total Cu	mg L^{-1}	0.0037^{+++}	0.0043	0.0159^{+++}	0.0010
Extractable Fe	mg L^{-1}	0.196	0.074	0.409^{+++}	0.045
Nelson River at the outlet of Split Lake					
Turbidity (Lab)	NTU	18.0^{+}	5.0	35.0^{+}	11.0^{+}
Extractable Al	mg L^{-1}	0.315^{+++}	0.126	0.651^{+++}	<0.100
Total Cu	mg L^{-1}	0.0050^{+++}	0.0075	0.0308^{+++}	0.0010
Extractable Fe	mg L^{-1}	$0.389^{+,+++}$	0.110	$0.672^{+,+++}$	0.129
Total Pb	mg L^{-1}	0.0009	0.0011	0.0066^{+++}	<0.0007

$^{+}$ Exceeds limit for drinking water.
$^{++}$ Exceeds limit for recreational use.
$^{+++}$ Exceeds limit for protection of aquatic life.

occasionally exceeded the aquatic life guideline at the outlet of Split Lake although these occasional high levels are not directly attributable to the CRD.

The consistent exceedence of the aquatic life guideline for extractable Al and total Cu, and of both the aquatic life and drinking water guidelines for extractable Fe in the northern drainages suggests a review of the applicability of these guidelines in the study area is in order. It remains to be determined if the naturally high levels of these variables are responsible for any impairment of drinking water quality or have any other ecological significance. If not, then it may be appropriate to develop specific water quality objectives for the area to accommodate these naturally higher levels.

4.2 Temporal Variation in Natural River Water Quality

The natural rivers (Burntwood River above Threepoint Lake, Footprint River above Footprint Lake) were distinguished by considerable stability in water quality between the 1971-1974 pre-development sampling period and the 1987-1989 FEMP study. Significant changes were detected in just 3 of the 17 variables for which comparable pre- and post-development data existed. Decreases in pH were detected in both the Footprint and Burntwood rivers. Decreases in NO_2-NO_3 and SO_4 occurred in the Footprint River and a marginally significant decrease in SO_4 occurred in the Burntwood River. All of these changes appear to have been due to alterations in sampling or analytical methods or to changes in the laboratories which conducted the analyses. Any comparable decreases in pH, SO_4, or NO_2-NO_3 at the sites affected by hydroelectric development were, therefore, attributed to methodological sources.

Long-term temporal trends in metals could not be examined in the natural rivers. Data on extractable or total metal species are essentially non-existent for the pre-development period. Several dissolved metals were routinely analyzed in the pre-development period, but the design of the FEMP study provided just one year of post-development measurements, eliminating the possibility of a statistical comparison.

No pre-development water quality data exist for the Aiken River, precluding an analysis of temporal trends at this site.

4.3 Changes in Water Quality Related to the Hydro Developments

4.3.1 Norway House Area

Impacts of LWR on water quality in the Nelson River below Sea River Falls and in the Jack River could not be assessed due to limited (Sea River Falls) or non-existent (Jack River) pre-development data records.

4.3.2 Nelson House and Threepoint Lake Area

The only adverse effect of CRD on water quality in the Rat River above Threepoint Lake was an approximately 3-fold increase in turbidity (Table 35) caused by bank and shoreline erosion on the Rat River below Notigi Dam and in the southern part of Notigi Reservoir. The primary source of this turbidity appears to be

Table 35. Water quality variables which changed significantly following Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1971 through 1974. X_{post} denotes the mean for the post-development years 1987 through 1989.

Variable	Units	X_{pre}	X_{post}
Rat River above Threepoint Lake			
Conductivity (Lab)	$\mu S\ cm^{-1}$	120	102
Turbidity (Lab)	NTU	5	17
Alkalinity (Total)	$mg\ L^{-1}$	56.4	46.4
Hardness (Total)	$mg\ L^{-1}$	61.2	45.9
TOC	$mg\ L^{-1}$	10.7	6.38
NO_2-NO_3	$mg\ L^{-1}$	0.237	0.019
Ortho PO_4	$mg\ L^{-1}$	0.005	0.002
Dissolved Na	$mg\ L^{-1}$	2.2	2.7
Dissolved K	$mg\ L^{-1}$	0.9	1.02
Dissolved Ca	$mg\ L^{-1}$	18.2	12.3
Dissolved Cl	$mg\ L^{-1}$	1.0	1.1
Dissolved F	$mg\ L^{-1}$	0.05	0.09
Reactive Si	$mg\ L^{-1}$	3.6	1.83

Notigi Reservoir.

Many other significant changes also occurred, although none can be classified as adverse. Annual mean conductivity, alkalinity (total), hardness (total), TOC, ortho PO_4, Ca, and reactive Si in the Rat River decreased by 15 to 60% between the 1972-1973 pre-development sampling period and the FEMP study (Table 35). Significant increases in Na, K, Cl, and F, by 10-80%, also occurred. All of these changes are directly attributable to diversion and are a reflection of differences in water quality between the Churchill River and the pre-existing Rat River. In most cases, the changes were no greater than the difference between the Rat and Churchill rivers, although the decrease in TOC was greater. This suggests some loss of TOC from the water column in passing through SIL and Notigi Reservoir. Decreases in SO_4 and NO_2-NO_3 were comparable to those in the natural rivers and were attributed to changes in analytical methodology.

The total P concentration in the Rat River did not change following diversion despite a lower concentration in the Churchill River. Release of P from flooded soils and vegetation probably accounts for this lack of a decrease.

Dissolved Fe, Mn, Cu, Pb, and Zn concentrations were not affected by diversion. The impact of diversion on other metal species is uncertain due to the lack of pre-development data.

Impacts of diversion on water quality in the Burntwood River at the outlet of Threepoint Lake could not be quantified due to the lack of pre-development data for this site. Significant changes would be expected and these should be qualitatively similar to those in the Rat River. However, the impact of CRD on turbidity in the outflow was probably much smaller than in the Rat River. Turbidity in the Rat River after diversion was the same as the natural level in the Burntwood River which accounted for about 60% of the outflow discharge prior to diversion.

4.3.3 Burntwood River at Thompson

The only adverse effect of CRD on water quality in the Burntwood River at Thompson is elevated turbidity due to bank and shoreline erosion. Many other changes in water quality occurred in addition to the increase in turbidity (Table 36), but none can be classified as an adverse effect based on the definition employed in this study. All of the other significant changes fall in the category of an improvement in water quality due to the introduction of softer Churchill River water.

Most of the changes in water quality did not occur until after the autumn of 1977, when the diversion was fully operational, although several were evident during CRD construction (from June, 1974, to August, 1976), when Rat River flows were cut off at the outlet of Notigi Lake, or during the immediate post-diversion period (September, 1976, to December, 1977), as flows were gradually increased. Turbidity and hardness increased during the period of CRD construction (Table 36). The increase in hardness was due to reduced flows on the Burntwood River during Notigi Dam construction and was restricted to the construction period. About

Table 36. Water quality variables in the Burntwood River at Thompson which changed significantly following Churchill River diversion. X_{pre} denotes the pre-development period from 1967 to May 1974. X_{const} denotes the construction period between June 1974 and August 1976. X_{post} is the immediate post-diversion period from September 1976 to December 1977. $X_{current}$ represents the FEMP study period between January 1987 and October 1989.

Variable	Units		
		X_{pre}	X_{const}
Turbidity (Lab)	NTU	14	23
Hardness (Total)	mg L^{-1}	69	77
		X_{pre}	X_{post}
Total P	mg L^{-1}	0.028	0.065
		X_{pre}	$X_{current}$
Conductivity (Lab)	μS cm^{-1}	141	115
Turbidity (Lab)	NTU	14	27
Alkalinity (Total)	mg L^{-1}	65	53
Hardness (Total)	mg L^{-1}	69	53
TOC	mg L^{-1}	12.63	6.94
Dissolved NO_2-NO_3	mg L^{-1}	0.140	0.022
Reactive Si	mg L^{-1}	3.94	2.01
Dissolved Ca	mg L^{-1}	21.3	14.6
Dissolved SO_4	mg L^{-1}	4.1	2.7
Dissolved Fe	mg L^{-1}	0.138	0.077
		X_{const}	X_{post}
Conductivity (Lab)	μS cm^{-1}	148	125
Alkalinity (Total)	mg L^{-1}	70	60
Hardness (Total)	mg L^{-1}	77	66
Total P	mg L^{-1}	0.039	0.065

Table 36. continued.

Variable	Units		
		X_{const}	$X_{current}$
Conductivity (Lab)	$\mu S\ cm^{-1}$	148	115
Alkalinity (Total)	$mg\ L^{-1}$	70	53
Hardness (Total)	$mg\ L^{-1}$	77	53
TOC	$mg\ L^{-1}$	14.00	6.94
Reactive Si	$mg\ L^{-1}$	4.10	2.01
Dissolved Mg	$mg\ L^{-1}$	6.5	4.0
Dissolved Ca	$mg\ L^{-1}$	21.5	14.6
Dissolved SO_4	$mg\ L^{-1}$	4.0	2.7
Extractable Mn	$mg\ L^{-1}$	0.037	0.021
Extractable Al	$mg\ L^{-1}$	0.870	0.488
		X_{post}	$X_{current}$
Turbidity (Lab)	NTU	18	27
Alkalinity (Total)	$mg\ L^{-1}$	60	53
Hardness (Total)	$mg\ L^{-1}$	66	53
TOC	$mg\ L^{-1}$	15.00	6.94
Total P	$mg\ L^{-1}$	0.065	0.036
Reactive Si	$mg\ L^{-1}$	3.25	2.01
Dissolved SO_4	$mg\ L^{-1}$	4.0	2.7
Extractable Fe	$mg\ L^{-1}$	0.786	0.628
Extractable Mn	$mg\ L^{-1}$	0.030	0.021

25% of the increase in turbidity during construction was due to elimination of inflow from the Rat River, with the balance of the increase possibly due to in-stream work and bank stabilization activities.

Total P concentrations approximately doubled between the construction and immediate post-impoundment periods (Table 36), probably due to release of P from flooded organic soils and vegetation. Pre-development levels were re-established before 1987 but this does not mean the P release has ended. The annual mean total P concentration in the Burntwood River is still more than twice that in the Churchill River near Leaf Rapids. Most of the P enters the water column as particulates; TDP at Thompson is 50% higher than in the Churchill River while particulate P is 300% higher.

Significant water quality changes between the pre-development and current (1987-1989) study periods included decreases in conductivity, alkalinity, hardness, NO_2-NO_3, reactive Si, and dissolved Ca, SO_4, and Fe, and an approximate doubling of turbidity (Table 36). Decreases in dissolved Mg and in extractable Al and Mn occurred between the construction and current study periods. The primary cause of the increase in turbidity was shoreline and bank erosion on the Burntwood River below Threepoint Lake. All of the decreases, with the exception of SO_4 and NO_2-NO_3, were due to the introduction of softer Churchill River water. The decreases in both SO_4 and NO_2-NO_3 were attributed to changes in analytical methods, laboratories, or both.

Concentrations of dissolved and extractable Al have not been elevated by CRD. A previous report of increased extractable Al following diversion was the product of the mistaken classification of pre-development dissolved Al data as extractable Al.

4.3.4 Split Lake Area

4.3.4.1 Burntwood River at Inlet to Split Lake

The changes water quality at this site caused by CRD undoubtedly were similar to those documented upstream at Thompson. No quantitative assessment of water quality changes at this site since CRD was possible due to the lack of pre-development data.

4.3.4.2 Nelson River at Inlet to Split Lake

Several significant differences in water quality were noted for the Nelson River above Split Lake between the 1971-1974 pre-development study period and the FEMP study (Table 37) but none of these changes could be directly attributed to LWR. The FEMP study was conducted during a period of very low Nelson River flows caused by drought conditions in western Canada. These changes could therefore be due in part, or in total, to the lower flows. There was no way of isolating the impact of LWR from any drought effects because the FEMP study comprised the entire post-development data record for the site.

Table 37. Water quality changes at sites in the Split Lake area following Lake Winnipeg regulation and Churchill River diversion. X_{pre} denotes the mean for the pre-development years 1972 and 1973. X_{post} denotes the mean for the post-development years 1987 through 1989. For modelled changes, X_{pre} is the current concentration in the Nelson River at the inlet to Split Lake.

Variable	Units	X_{pre}	X_{post}
Nelson River at the inlet to Split Lake			
pH (Lab)	units	8.24	7.85
NFR	mg L^{-1}	15	5
Dissolved K	mg L^{-1}	2.43	2.06
TDP	mg L^{-1}	0.020	0.011
TDN	mg L^{-1}	0.388	0.304
Reactive Si	mg L^{-1}	0.72	1.24
Nelson River at the outlet to Split Lake			
Based on a pre/post comparison			
pH (Lab)	units	8.06	7.81
Conductivity (Lab)	μS cm^{-1}	307	228
Hardness (Total)	mg L^{-1}	120.0	91.7
Dissolved Ca	mg L^{-1}	33.9	24.1
Dissolved K	mg L^{-1}	2.26	1.65
Dissolved Na	mg L^{-1}	14.3	9.3
Dissolved Cl	mg L^{-1}	15.3	9.5
Dissolved SO_4	mg L^{-1}	26.0	13.7
Based on modelled changes			
Dissolved As	mg L^{-1}	0.0007	0.0005
Dissolved Fe	mg L^{-1}	0.193	0.387
Extractable Mn	mg L^{-1}	0.010	0.015

4.3.4.3 Nelson River at Outlet of Split Lake

An increase in extractable Fe, such that the annual mean concentration at the outlet now exceeds the Canadian Water Quality Guidelines for drinking water and protection of aquatic life by 30% (Table 37), has been the only adverse effect of CRD on water quality in Split Lake. This increase, and significant decreases in TDS, DIC, and dissolved As were inferred on the basis of current differences in concentration between the Burntwood and Nelson rivers and the relative contribution of these rivers to total inflow.

Churchill River diversion has not caused a significant increase in annual mean water turbidity or suspended sediment concentration (NFR) or a decrease in water transparency despite the much higher suspended sediment concentrations in the Burntwood River than in the Nelson River. Much of the suspended sediment load of the Burntwood River appears to settle out near the mouth, preventing a general increase in suspended sediment levels in the lake or downstream. Sediment deposition also appears to have prevented increased concentrations of extractable Al and total Ni and V in Split Lake.

Statistical analysis of pre- and post-development data for 24 variables revealed significant declines had occurred in 8 (Table 37). All but the decline in pH were a direct result of diversion. The decrease in pH at the Split Lake outlet is consistent with the decrease which occurred in the Nelson River near the inlet and in the natural rivers during the same period, indicating the change is not related to the hydroelectric developments. None of these significant changes based on pre/post comparisons represents an adverse effect.

4.4 Water Quality - Water Discharge Relationships

The natural rivers, sites on the Nelson River upstream of Split Lake, and the Burntwood River at Thompson prior to diversion were characterized by having many discharge-correlated variables, with negative correlations dominating. Negatively correlated variables generally included the major cations (Na, K, Ca, Mg) and anions (Cl, F, SO_4) and associated variables (conductivity, alkalinity, hardness, TDS), some metals, DIC, reactive Si, and dissolved N species (TDN, NO_2-NO_3, total NH_3). Positive correlates at these sites generally included turbidity, NFR, colour, and the organic carbon species (DOC and TOC).

The discharge relationships were profoundly different at sites on the CRD. In the Rat River above Threepoint Lake and in the Burntwood River at the outlet to Threepoint Lake, at Thompson, and at the inlet to Split Lake all correlations with discharge were positive following diversion. This is evidence of the increased importance of eroded shoreline materials in regulating water quality. Whereas water quality typically was highest during periods of peak flow prior to diversion, water quality now decreases with increasing river flows.

The discharge relationships also were weaker, in general, at the sites on the CRD than in the natural rivers. This appears to be a product of the less dynamic discharge regime following diversion. Although mean discharge on the Rat and Burntwood rivers was substantially elevated due to the diversion, the variation in discharge relative to the mean was an order of magnitude lower.

It is unclear if LWR has had any effect on water quality - water discharge relationships in the Nelson River at the inlet to Split Lake or if either LWR or CRD has affected the relationships at the outlet of Split Lake. The relationships during the FEMP study differed from those which prevailed prior to development, and the changes at the outlet to Split Lake were similar to those which occurred at sites on the CRD, but the effect of the hydroelectric developments could not be separated from effects of the low Nelson River flows during the FEMP study.

The water quality - water discharge relationships derived from the FEMP study data offer limited potential for impact prediction or management. The strongest correlations were found at sites where there is the least need for their application; the natural rivers and the Nelson River at Sea River Falls. Most relationships at sites on the CRD were too weak (i.e., $r^2 < 0.25$) to have much predictive ability. Several important exceptions included the relationships for NFR and turbidity in the Burntwood River at Thompson, and the relationships for turbidity and alkalinity in the Burntwood River at the inlet to Split Lake. None of the water quality - water discharge relationships for the Nelson River at the inlet or outlet to Split Lake should be used for impact prediction or management purposes until the influence of the low Nelson River flows on these relationships can be determined.

4.5 Seasonal Variations in Water Quality

A number of distinct seasonal trends were evident in the comparison of water quality under ice cover and in the open water season during the 1987-1989 study period. Lower chlorophyll a concentrations occurred under ice cover at several sites (Jack R., Rat R. above Threepoint L., Burntwood R. at Thompson, and Burntwood R. at the inlet to Split Lake). This may be a result of the imposition of light limitation on algal growth by accumulated snow and ice cover. Elevated NO_2-NO_3 concentrations under ice-cover also occurred at most sites (Rat R. above Threepoint Lake, Burntwood R. at the inlet and at the outlet of Threepoint Lake and at the inlet to Split Lake, and the Nelson R. at the inlet and outlet to Split Lake). This could be a result of lower photosynthetic demand for nitrogen at the lower light levels and cooler temperatures under ice cover. At some sites (Rat R. above Threepoint Lake, Burntwood R. at the outlet of Threepoint Lake, Nelson R. at the outlet of Split Lake), the higher NO_2-NO_3 levels under ice contributed to higher TDN and, in some cases (Rat R.), higher total N concentrations.

A variety of other variables displayed marked seasonality at specific sites, with a larger number of seasonally variant parameters in the smaller natural rivers (i.e., Footprint and Aiken rivers) than in the larger Burntwood, Rat, and Nelson rivers. These variables included major ions and associated variables like conductivity, alkalinity, hardness and in some cases metals and NFR. Most of these variables increased under ice cover, although NFR typically was lower in winter. For variables which were correlated with discharge, these seasonal differences were accounted for by seasonal variations in river discharge.

At sites where water quality measurements were made prior to development, samples under ice cover generally were much less frequent than in the FEMP study. This restricted the examination of changes in seasonal patterns related to the hydroelectric developments to a qualitative analysis. The analysis was further restricted by the few variables which were common to the pre- and post-development periods at many sites. Although several changes in seasonal patterns were noted, none could conclusively be attributed to the hydroelectric developments. The most complete analysis was possible for the Burntwood River at Thompson, where measurements were routinely made under ice-cover in the pre-development and construction periods. Only chlorophyll a showed marked seasonality in any of the pre-development, construction, or current study periods.

5.0 Impact Mitigation and Management Options

5.1 Burntwood River - Threepoint Lake to Split Lake

The major impact of CRD on this reach of the Burntwood River has been the increase in turbidity due to shoreline and bank erosion along the Burntwood River and upstream on the Rat River and in southern Notigi Reservoir. Elevated turbidity also is the only adverse water quality impact currently evident in this section of the CRD. Natural bank stabilization processes should eventually contribute to a reduction in turbidity, but the length of time this will require and the level of recovery which will be achieved are difficult to predict.

5.2 Split Lake

The development of measures to mitigate adverse effects of CRD on Split Lake water quality does not appear to be justifiable at this time. Notwithstanding the several significant changes in water quality related to CRD, the only adverse effect of the diversion on water quality which has been documented thus far is the increase in mean extractable Fe above the drinking water and aquatic life guidelines.

Although the guidelines are now exceeded in Split Lake, there is no reason to believe that drinking water quality or aquatic life have actually been adversely affected. Naturally high Fe levels are prevalent in the study area. However, should information become available in the future which suggests significant environmental degradation has occurred as a result of this increase in Fe, requiring mitigation at source rather than treatment before consumption, this should be possible through management of the relative contribution of Burntwood River flows to Nelson River flows. A 50% reduction in Burntwood River flows should reduce the concentration in Split Lake to the guideline value.

6.0 Recommendations for Further Study

6.1 Norway House Area

The FEMP sampling program adequately characterized current water quality at these sites and represents a satisfactory baseline for assessment of the impact of any future hydroelectric projects. The impact of LWR on water quality at these sites could not be determined due to the shortage of pre-development data. This deficiency cannot be alleviated by further sampling.

It may be possible to infer the nature of any water quality changes which have occurred at the Jack River or Sea River Falls sites by comparing the FEMP study data with existing pre-development data for upstream sites. Water quality in the Nelson River at Warren Landing, sampled since 1969, might be a suitable surrogate for the Jack River. Similarly, data for the Nelson River at Norway House might be a surrogate for Sea River Falls. In order to use pre-development data from these surrogate sites, it must be determined that there are, or were, no natural differences in water quality between the FEMP sites and their respective surrogates or at least that any natural differences between the sites were not similar in character to those which might be caused by the hydroelectric development. Furthermore, this validation of the surrogate sites must be done indirectly, using information on natural erosion sites and locations of man-made facilities, due to the paucity or complete lack of pre-development data for the FEMP sites.

6.2 Burntwood River - Threepoint Lake to Split Lake

The only adverse impact of Churchill River diversion on water quality in the Burntwood River system has been the increase in turbidity due to bank and shoreline erosion. The higher turbidity in the Rat River appears to be from Notigi Reservoir while the increase in the Burntwood River is primarily due to erosion occurring Threepoint Lake. Although this change can be classified as an adverse impact on the basis of the Canadian Water Quality Guidelines, and the general nature of any associated ecological changes can be stated on the basis of studies conducted elsewhere, any specific impacts of increased turbidity on the ecology of the Burntwood River remain largely unknown, as most studies have focused on lakes in the Rat River valley. Investigation of the ecological effects of increased turbidity in the system will be necessary for the establishment of water quality objectives to be met by hydroelectric generating stations which are constructed on the Churchill-Nelson system in the future.

Unfortunately, any assessment of biological impacts will be more limited by the availability of pre-development data than was the analysis of water quality impacts presented in this report. Nevertheless, some review of biological impacts related to increased turbidity on the Burntwood River is warranted, if only to provide adequate documentation of the limitations of any such analysis.

The overall impact of elevated turbidity is determined by both the magnitude and duration of the increase. This report documents magnitude, but none of the data gathered to date provide an indication of duration. The eroding banks and shorelines should gradually stabilize over time, but the lack of a significant decline in turbidity since diversion indicates this may be a very slow process. An estimate of duration is necessary both for complete documentation of the impact of CRD on the sediment regime in the Burntwood River, as well as for assessment of the impact of future modifications to the operation regime.

There is no need for continued broad-spectrum water quality monitoring on the Burntwood River for assessment of the major impacts of CRD. Additional data collection will never make up for deficiencies in the pre-development database and, unless there is a change in the way the diversion is managed, there is little reason to expect further substantial changes in water quality.

Although the major changes in water quality appear to have occurred, the length of time required for the system to reach a new equilibrium is unknown. Thus there is strong justification for the continued monitoring of water quality at a minimum of one site. Located below the major reservoirs on the CRD, the Thompson site is well-placed to examine long-term changes in water quality on the CRD. Other sites might be considered, but the Thompson site has the most complete data record of any site in the Churchill-Nelson system and is readily accessible. Annual monitoring is probably unnecessary, with a 5 year interval between monitoring periods providing adequate coverage. Each monitoring period should be a 2-year study which repeats the FEMP study sampling and analytical methods exactly. A number of variables could be dropped from the program to reduce cost without losing important information. Deleted variables could include all those which were consistently at or below the limit of detection, including total Ba, dissolved and total Cd, total Co, and total Hg. Other variables which might be dropped include the field measurements of pH, conductivity, and turbidity because all of the pre-development data are laboratory values.

Any future field activities should include an examination of the predictive ability of the water quality - water discharge relationships identified in the present study. Such studies would involve the measurement of discharge-correlated variables over a range of discharges, and the comparison of predicted and observed values.

6.3 Split Lake

Contrary to the perception of some members of the Split Lake community, this and other studies have shown CRD has not caused a general increase in turbidity or other suspended sediment-related variables in Split Lake. This does not necessarily indicate the perception of increased turbidity is incorrect, only that any increase is not widespread. The diversion has undoubtedly had a substantial impact on sediment deposition rates in Split Lake near the mouth of the Burntwood River, and possibly has affected sediment chemistry in this region as well as causing a local elevation in turbidity. Documentation of the extent and magnitude of sediment-related impacts in this part of the lake is required. Variables monitored should include the rate and location of sediment deposition and the chemical composition of the sediments with emphasis on metals and factors regulating metal mobilization.

The FEMP and other sampling programs which have been conducted in Split Lake examined changes in turbidity and other variables in the central basin and outlet. This approach was necessary to detect any major, wide-spread, changes but could easily have missed localized impacts or changes around the perimeter of the lake. The higher lake level since diversion does not appear to have caused widespread shoreline erosion, as expected since the increase was within the natural range of lake levels. It is not known if any isolated or localized increases in erosion have occurred, and this possibility should be investigated. This might be done using remote sensing techniques accompanied by appropriate ground-truthing surveys.

The increase in extractable Fe in Split Lake is the only change in water quality which can be categorized as adverse, yet it is unlikely that this increase has had an effect on drinking water quality or on aquatic life. The prevalence of sites in the NFA study area where extractable Fe naturally exceeds the Canadian water quality guidelines for consumption and protection of aquatic life suggests the guidelines may be inappropriate for this area. A more complete review of information on the effect of Fe on drinking water quality and aquatic life than was possible in this report should be conducted to assess the applicability of the guideline in the area. Additional monitoring of extractable Fe levels in Split Lake may be required if the current elevated levels are shown to have impaired water quality.

The low Nelson River flows during the FEMP study period made it difficult to determine if the apparent changes in water quality at the inlet to Split Lake, and in the water quality - water discharge relationships at the inlet and outlet to Split Lake, were due to the hydroelectric developments or to low flow conditions. There is a need for continued monitoring over a wider range of flows to discriminate natural from project-induced changes, if any have occurred.

The monitoring program should be consistent with the design of the FEMP study, with a similar sampling frequency, although a number of the variables measured in the FEMP study could be dropped from any subsequent program in order to minimize cost without any significant reduction in information, as identified above. A two year program would be the minimum to permit a statistical analysis.

6.4 Bacteriological Studies

The bacteriological studies in recreational waters conducted under the FEMP were limited in the scope of their conclusions due to non-compliance with the sampling requirements specified in the water quality guidelines at most of the study sites. The need for complete documentation of recreational water quality compliance with the fecal coliform guideline should be reviewed with the appropriate responsible agencies and the communities to determine if additional sampling, according to the guideline protocol, is required.

6.5 General Recommendations

The results of the FEMP water quality study indicate the major changes in water quality caused by the Churchill-Nelson hydroelectric development have been increases in turbidity due to shoreline erosion, the release of phosphorus from flooded materials, the loss of organic carbon from the water column, and changes in water quality related to the mixing of water masses from different sources. These features should be considered in the design of monitoring programs for assessment of the effects of additional generation projects in the system. The requirements of a monitoring program for a specific generating station will be very dependent on its location, the proposed extent of flooding, and the operation regime.

The FEMP water quality study placed considerable emphasis on the measurement of metals. This emphasis could be reduced considerably in any future studies. Few changes in metals concentrations have occurred in the Churchill-Nelson system due to hydroelectric development, and then only as a result of the mixing of water masses with naturally different metals concentrations. There is no evidence of a general mobilization of metals from eroded soils and vegetation.

Unfortunately, the total mercury data could not be used because the limit of detection of the analytical method was well in excess of the demonstrated actual concentrations and some apparent sample contamination. Should it be considered necessary to measure total mercury concentrations in water in the future, the sampling program should be designed to measure levels in the 0.5 to 3.0 ng L^{-1} concentration range. This will require a dedicated mercury sampling program which employs clean sampling technique and an appropriate analytical method. The utility of any program to measure total mercury levels in water is questionable, given that other studies have shown the elevated mercury levels in fish from flooded lakes along the CRD are in no way related to total mercury concentrations in water or sediments, that total mercury levels in water are not elevated by impoundment, and that natural levels are well below current drinking water standards.

The number and location of stations sampled in the FEMP study appear to have been sufficient for documentation of the impacts at the sites which have been developed to date, covering the major components of the system. Additional stations would not improve the assessment due to the absence of pre-development data. The data collected to date also provide a substantial baseline for assessment of the impacts of future projects although additional stations may be required to examine specific or localized impacts. It would be premature to identify additional station locations at this time without knowledge of the specific nature of future projects. Three years of study prior to construction, with a sampling frequency comparable to that of the FEMP study, would be required to establish pre-development baseline conditions at any new sampling station.

The results of the FEMP study indicate there is really no substitute for solid pre-development data. The natural rivers were somewhat useful for examining natural levels of change in the few variables which were common to the pre- and post-development periods, but were of no use for the filling in of data gaps at the regulated sites. The pre-development databases for the natural rivers were just as limited as for the regulated sites. The Aiken River data were of no use in this regard as there were no data from the pre-development period.

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8.0 Glossary of Technical Terms, Chemical Symbols, and Units of Measure

Al: Chemical symbol for the element aluminum.

As: Chemical symbol for the element arsenic.

Ba: Chemical symbol for the element barium.

Biocide: a substance which is destructive to many different organisms (e.g., DDT).

Ca: Chemical symbol for the element calcium.

Cd: Chemical symbol for the element cadmium.

Chlorophyll a: A photosynthetic pigment in plants. The concentration is used as a measure of algal biomass.

Cl: Chemical symbol for the element chlorine.

Co: Chemical symbol for the element cobalt.

Coefficient of variation (c.v.): Calculated as the ratio of the standard deviation to the mean, expressed as a percentage. A measure of the degree of variation in an observed quantity (e.g., river discharge).

CRD: Churchill River diversion.

Cu: Chemical symbol for the element copper.

Detection limit: The smallest concentration or amount of a substance which can be reported as present with a specified degree of certainty by a definite, complete, analytical method.

DIC: Dissolved inorganic carbon.

Dissolved metal: A metal constituent in an unacidified sample that passes through a 0.45 μm membrane filter.

EC_{50}: Median effective concentration. The concentration of material in water to which test organisms are exposed that is estimated to be effective in producing some sublethal response in 50% of the test organisms. The EC_{50} is usually expressed as a time-dependent value (e.g., 24-h or 96-h EC_{50}).

Extractable metal: The concentration of a metal in solution after treatment of an unfiltered sample with hot dilute mineral acid. Applies to a method which will remove most but not all of a metal from a sample.

F: Chemical symbol for the element fluorine.

FEMP: Federal Ecological Monitoring Program.

Fulvic acid: One of three classes of **humic substances** defined as those substances which are soluble in both acid and base.

Guideline (water quality): A numerical concentration limit or narrative statement which has been established to support and maintain a designated water use.

Hg: Chemical symbol for the element mercury.

Humic acid: One of three classes of **humic substances** defined as those substances which precipitate with acidification.

Humic substance: Form the majority of organic matter in soil and water. Consist of dark-coloured and acidic compounds with molecular weights ranging from hundreds to thousands of daltons. Resistant to microbial degradation, these substances tend to persist in aquatic systems.

K: Chemical symbol for the element potassium.

LWCNR: Lake Winnipeg and Churchill and Nelson Rivers Project.

LWR: Lake Winnipeg regulation.

Mg: Chemical symbol for the element magnesium.

Mn: Chemical symbol for the element manganese.

N: Chemical symbol for the element nitrogen.

Na: Chemical symbol for the element sodium.

NFA: Northern Flood Agreement.

NFC: Northern Flood Committee.

NFR: Non-filterable residue. A measure of the concentration of suspended sediments in water. Defined as that mass of material in a volume of water which does not pass through a 0.45 μm membrane filter.

NH_3: Chemical symbol for the compound ammonia.

Ni: Chemical symbol for the element nickel.

NO_2-NO_3: Chemical symbol used to identify the compounds nitrite (NO_2) and nitrate (NO_3).

ortho PO_4: Phosphorus occurring as the compound PO_4^{3-}. Considered to represent the concentration of inorganic soluble phosphorus, the form directly available to aquatic plants.

P: Chemical symbol for the element phosphorus.

Pb: Chemical symbol for the element lead.

pH: Value which represents the acidity or alkalinity of an aqueous solution. It is defined as the negative logarithm of the concentration of hydrogen ions in a solution.

POC: Particulate organic carbon.

Primary production: The production of organic matter from inorganic material by organisms.

Q: Symbol used to represent river discharge.

Range: The difference between the lowest and highest value in a set of data.

React Si: Soluble reactive silica.

Replicate: A repeated measurement of a variable under specific conditions.

Se: Chemical symbol for the element selenium.

Si: Chemical symbol for the element silicon.

SIL: Southern Indian Lake.

SO_4: Chemical notation for the compound sulphate which consists of 1 sulphur and 4 oxygen atoms.

Sublethal: Involving a stimulus below the level which causes death.

Suspended metal: Those metals in an unacidified sample which are retained on a 0.45 μm membrane filter.

TDN: Total dissolved nitrogen.

TDP: Total dissolved phosphorus.

TDS: Total dissolved solids.

TIC: Total inorganic carbon.

TKN: Total Kjeldahl nitrogen. Total concentration of organic nitrogen and ammonia nitrogen.

TOC: Total organic carbon.

Total metal: The concentration of a metal in an unfiltered sample that is digested in strong nitric acid. Refers to a method which removes all traces of a metal from a sample for analysis.

True colour: The colour of water resulting from substances which are totally in solution: not to be mistaken for apparent colour resulting from colloidal or suspended matter.

V: Chemical symbol for the element vanadium.

Zn: Chemical symbol for the element zinc.

Appendix 1: Bacteriological Study Methods

Sampling Site Descriptions and Locations

Seasonal bacterial studies were conducted in the vicinity of the communities of Norway House, Cross Lake, Nelson House, Split Lake, and York Landing. Seasonal study site numbers and descriptions are listed in Tables A1, A2, A3, A4, and A5 and site locations are identified in Figures A1, A2, A3, A4, and A5. Intensive bacterial studies were conducted near all communities except York Landing. Intensive study site numbers and descriptions are listed in Tables A6, A7, A8, and A9 and site locations are identified in Figures A6, A7, A8, A9, A10, A11, A12, and A13.

Sampling and Analytical Methods

Water samples for fecal coliform determination were collected from shoreline beach areas near the NFA communities by wading into the water, or by boat in deeper embayment areas. Subsurface (0.5 m depth) water samples were collected in 1 L wide-mouthed autoclaved glass jars. The mouth of the sample was pointed in the upstream direction of the waterbody to eliminate any potential contamination from the sample collector, then immediately sealed. Samples were placed in ice-cooled coolers or trays and transported to the Water Quality Branch facilities in Thompson, Manitoba, for analysis.

Fecal coliform bacterial densities in the water samples were determined using the membrane filter (MF) procedure, as described in APHA (1985), with the following modifications. Known volumes of the raw water samples were vacuum filtered through 47 mm diameter, 0.45 μm pore-size, HC type, membrane filters. Sample volume was determined from site observations. Three volumes, ranging from 1 to 100 mL, were normally filtered for each station. Samples were incubated at 44.5 °C for 48 hours. Incubations commenced within 6 hours of sample collection. The results of the 3 bacterial counts for each sample were standardized to a 100 mL volume. Reported values are the arithmetic mean of the 3 counts.

Table A1. Bacteriological water quality sampling sites, Norway House, Manitoba, 1987. See Figure A1 for site locations.

	Site Number	Site Description
Seasonal Study	1	Green Dock adjacent to Band Office
	2	School Dock
	3	Government Dock, east of STP outfall
	4	Rocky Place near graveyard
	5	Asiniwapiska, "where the rocks are"
	6	Sewage Outlet Bay at Church Point
	7	Bridge into Fort Island
	8	Historic Dock by Bay Store Bridge
	9	Water intake

STP Sewage Treatment Plant

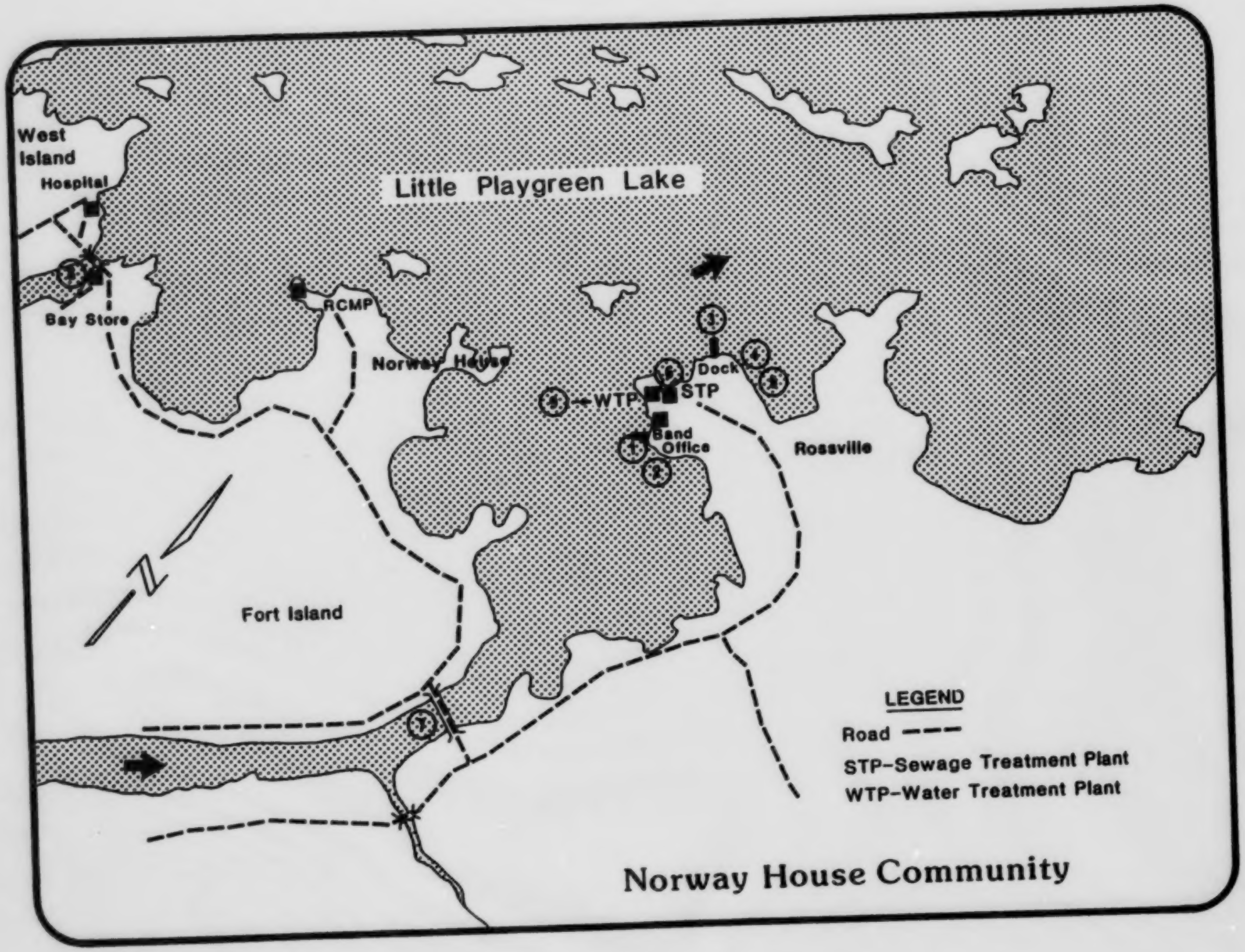

West Island
Hospital
Little Playgreen Lake
Bay Store
RCMP
Norway House
WTP
STP
Dock
Band Office
Rossville
N
Fort Island
LEGEND
Road ———
STP–Sewage Treatment Plant
WTP–Water Treatment Plant
Norway House Community

Figure A1. Location of 1987 seasonal bacterial study sampling sites near the community of Norway House, Manitoba. See Table A1 for site descriptions.

Table A2. Bacteriological water quality sampling sites, Cross Lake, Manitoba, 1987. See Figure A2 for site locations.

	Site Number	Site Description
Seasonal Study	1	Saggitawak, swimming area near Roman Catholic church
	2	Beardy's Point, swimming area near Saggitawak Bridge
	3	Saggitawak Bridge
	4	Wapak swimming area, near Junior High School, power line, and sewage area
	5	Natimeek Water Treatment Bay, to the right of water intake
	6	Natimeek - Robinson's Point, Water Treatment Bay to the left of the water intake
	7	Sand Bay
	8	Catherine's Point

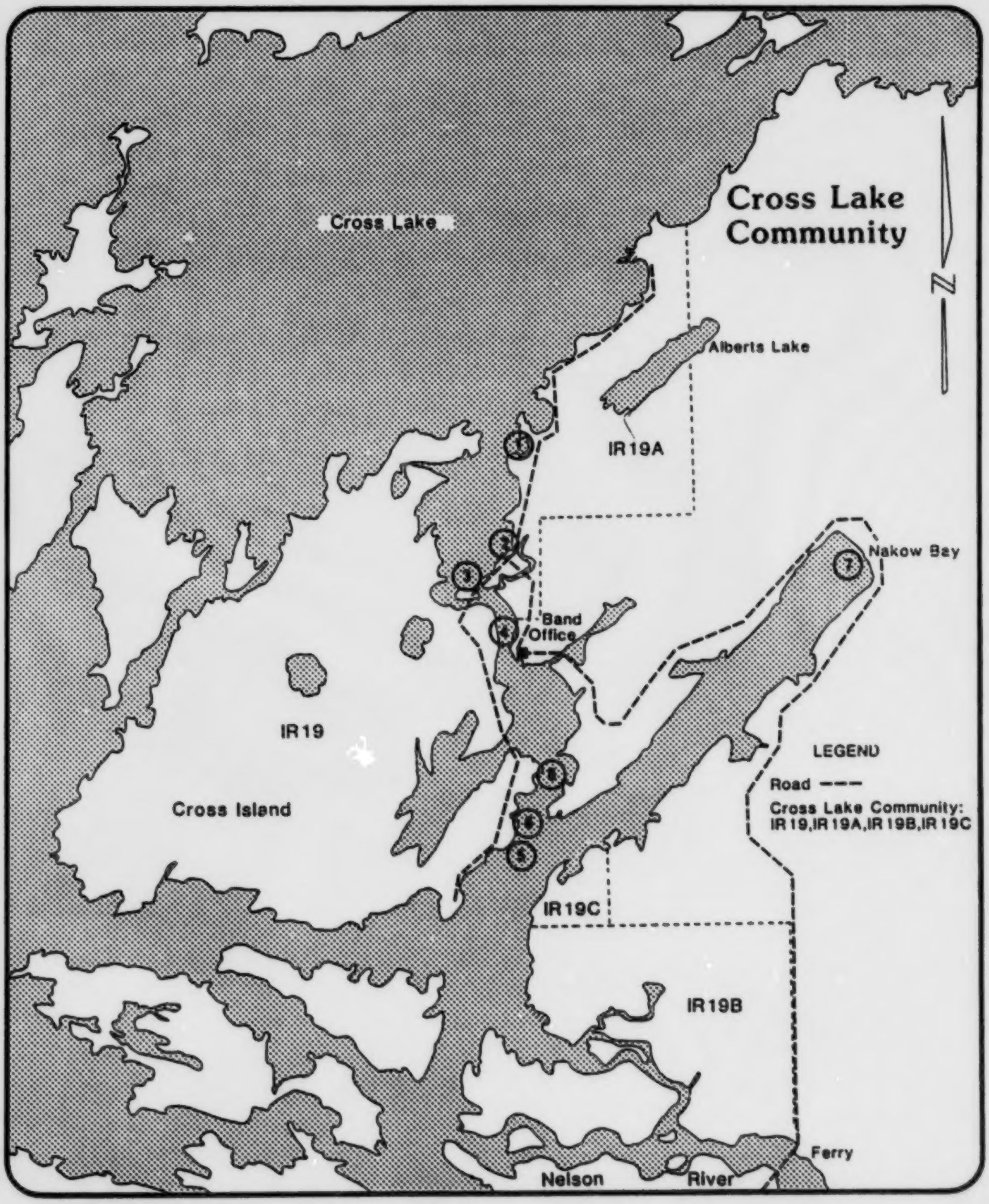

Cross Lake
Cross Lake Community
N
Alberts Lake
IR19A
1
2
3
Nakow Bay
7
Band Office
4
IR19
LEGEND
Road ---
Cross Lake Community: IR19,IR19A,IR19B,IR19C
Cross Island
8
6
5
IR19C
IR19B
Ferry
Nelson River

Figure A2. Location of 1987 seasonal bacterial study sampling sites near the community of Cross Lake, Manitoba. See Table A2 for site descriptions.

Table A3. Bacteriological water quality sampling sites, Nelson House, Manitoba, 1987. See Figure A3 for site locations.

	Site Number	Site Description
Seasonal Study	1	Metis Beach
	2	New Development Area, across from cemetary
	3	Church Bay (East Ramp), nearest the church
	4	Church Bay (West Ramp), furthest from the church
	5	Industrial Bay, by Causeway #2 near old industrial area
	6	School Bay, at outlet of sewage effluent
	7	Nursing Station Bay, near dock by pumphouse
	8	Inspiration Point, near dock, west of Nursing Station Bay
	9	Government Dock, most popular dock, just west of Bay Store, below the embankment
	10	Michael Bay
	11	WTP, raw water inside pumphouse
	12	Dog Point, bay east of Industrial Bay at Causeway #1

WTP Water Treatment Plant

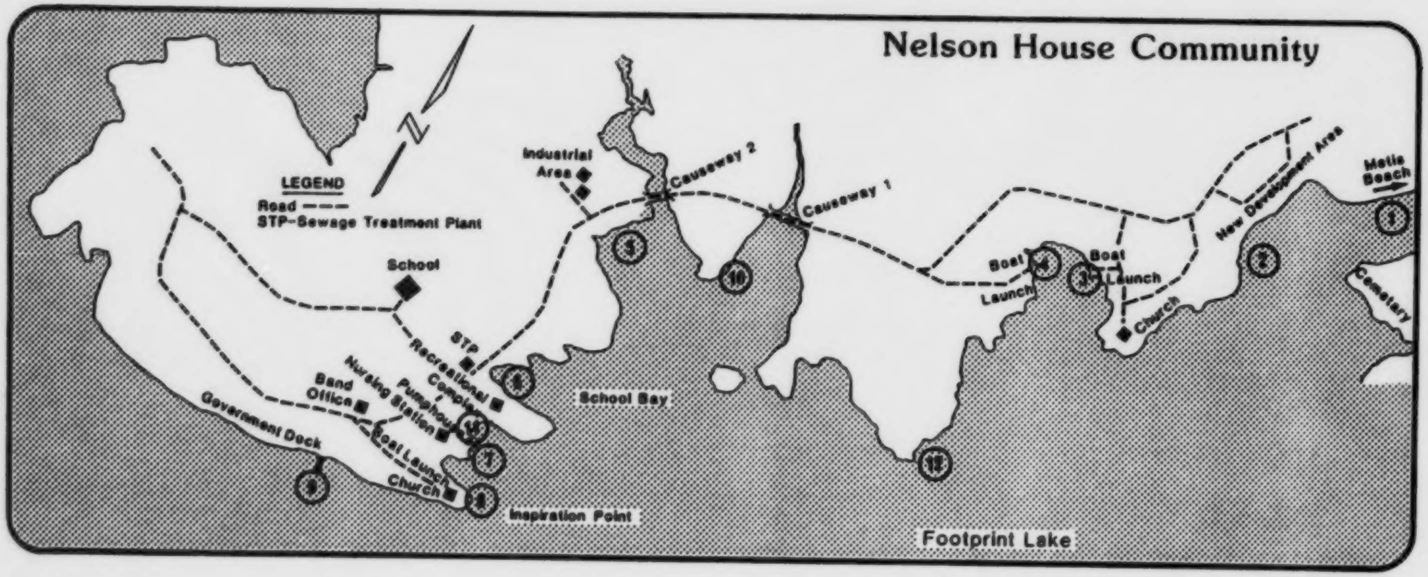

Nelson House Community
LEGEND
Road ---
STP-Sewage Treatment Plant
N
School
Industrial Area
Causeway 2
Causeway 1
STP
Recreational Complex
Band Office
Nursing Station
Pumphouse
Boat Launch
Church
Government Dock
Inspiration Point
School Bay
Boat Launch
Boat Launch
Church
New Development Area
Metis Beach
Cemetary
Footprint Lake
1
2
3
4
5
6
7
8
9
10
11
12

Figure A3. Location of 1987 seasonal bacterial study sampling sites near the community of Nelson House, Manitoba. See Table A3 for site descriptions.

Table A4. Bacteriological water quality sampling sites, Split Lake, Manitoba, 1987. See Figure A4 for site locations.

	Site Number	Site Description
Seasonal Study	1	Nursing Station Bay, opposite Nursing Station
	2	Bay Store Beach Area, at end of Bay Store back road
	3	First Beach, adjacent to Sewage outfall
	4	Beach Point, between Nursing Station and Bay Store
	5	Second Beach, near Ferry Dock
	6	School A (1)
	7	School B (3)
	8	WTP, treated water
	9	WTP, untreated water
	10	Water Treatment Plant Beach, east of old plant
	11	Water Treatment Bay, west of old plant
	12	Sewage Treatment Bay, near First Beach

WTP Water Treatment Plant

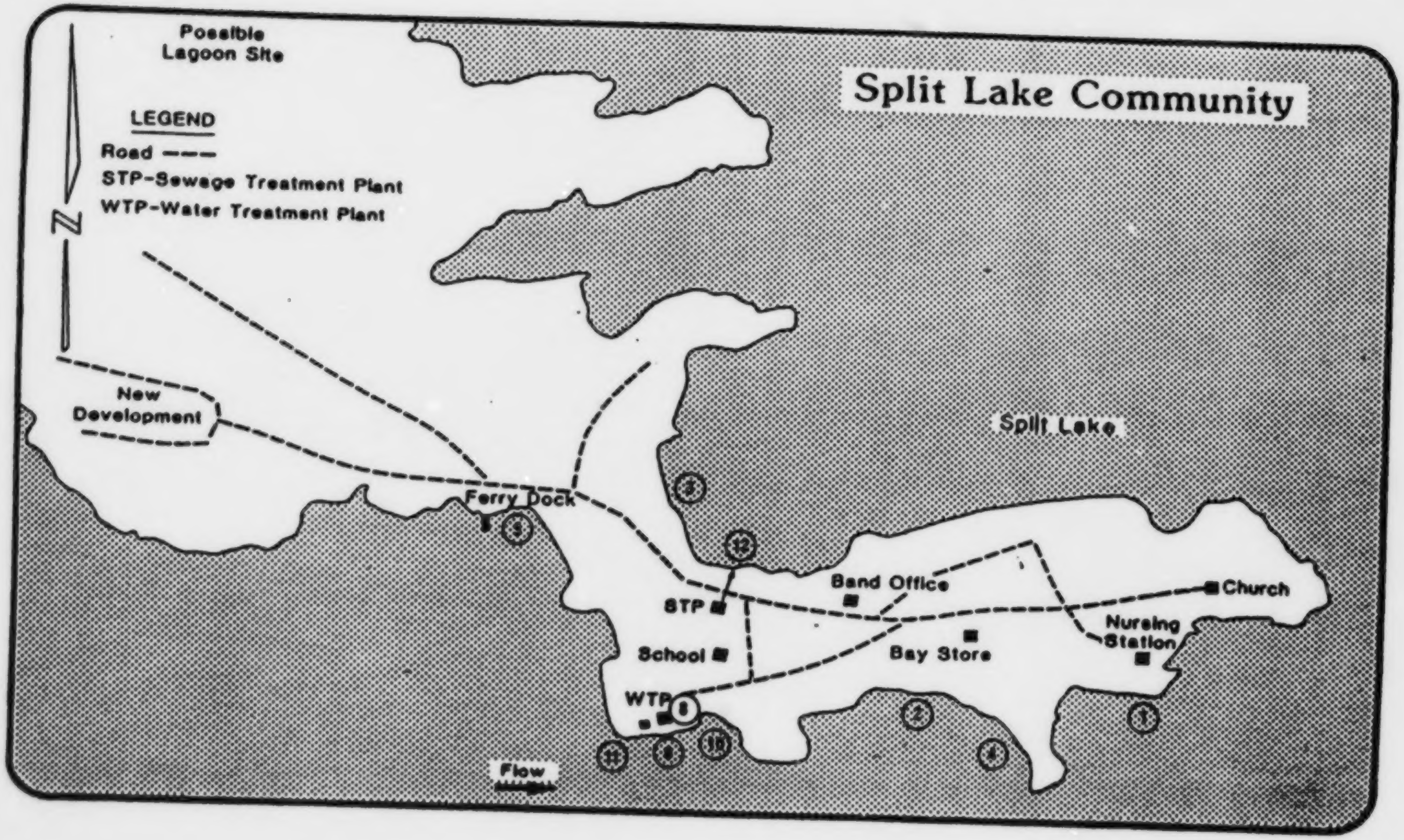

Split Lake Community
Possible Lagoon Site
LEGEND
Road ---
STP-Sewage Treatment Plant
WTP-Water Treatment Plant
N
New Development
Ferry Dock
Split Lake
Band Office
STP
School
WTP
Bay Store
Nursing Station
Church
Flow
1
2
3
4
5
8
9
10
11
12

Figure A4. Location of 1987 seasonal bacterial study sampling sites near the community of Split Lake, Manitoba. See Table A4 for site descriptions.

Table A5. Bacteriological water quality sampling sites, York Landing, Manitoba, 1987. See Figure A5 for site locations.

	Site Number	Site Description
Seasonal Study	1	Gravel Pit
	2	Water Treatment Plant holding tank
	3	Ferry Dock, east side
	4	Drainage Ditch outflow into river on west side of Ferry Dock
	5	Island Beach
	6	Sandy Beach
	7	Bearass Beach

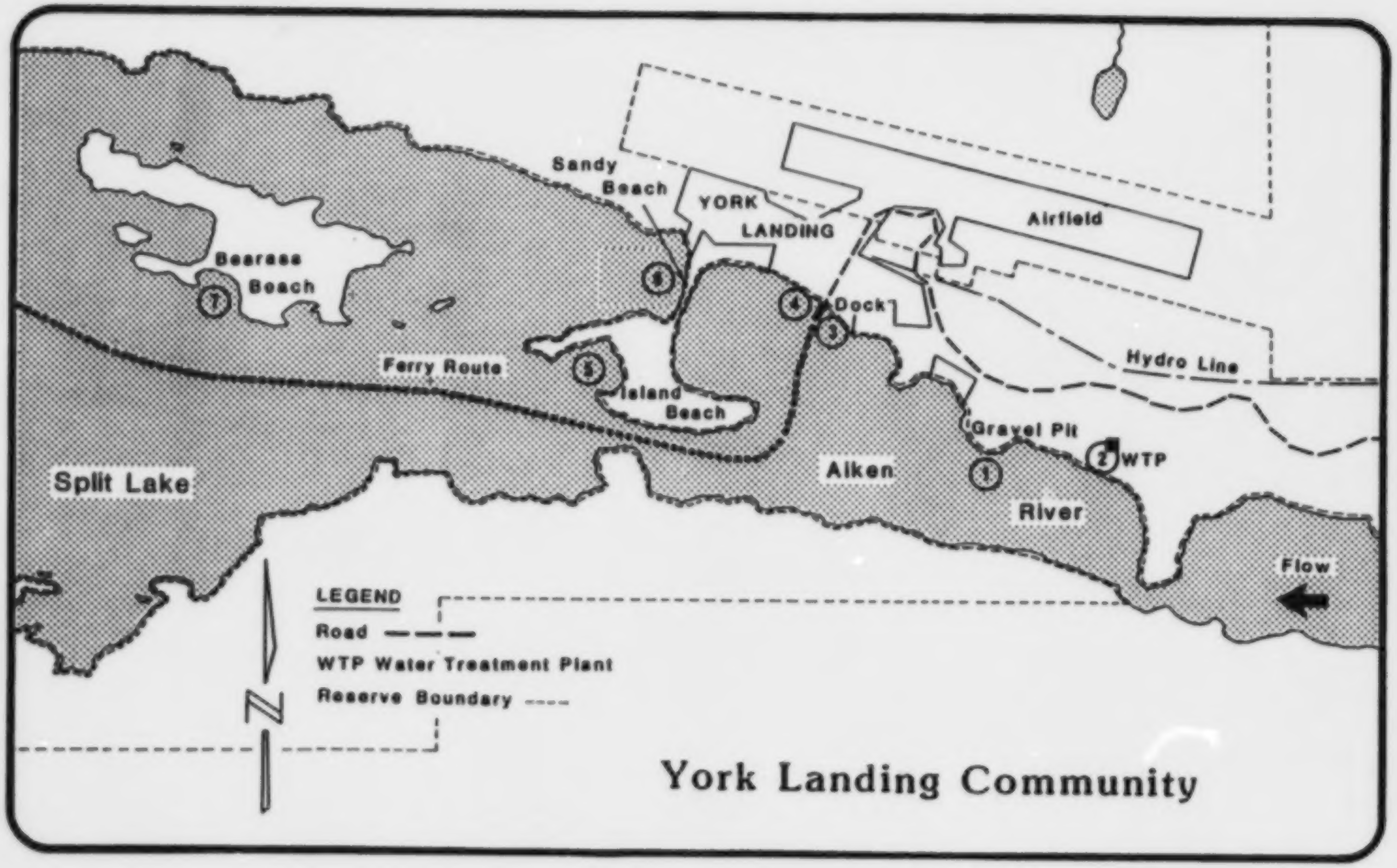

Sandy Beach
YORK LANDING
Airfield
Bearsse Beach
7
6
4
Dock
3
Ferry Route
5
Island Beach
Hydro Line
Gravel Pit
2 WTP
1
Split Lake
Aiken
River
Flow
LEGEND
Road
WTP Water Treatment Plant
Reserve Boundary
N
York Landing Community

Figure A5. Location of 1987 seasonal bacterial study sampling sites near the community of York Landing, Manitoba. See Table A5 for site descriptions.

Table A6. Bacteriological water quality sampling sites, Norway House, Manitoba, 1987. See Figures A6 and A7 for site locations.

	Site Number	Site Description
Intensive Study August 11	1	10 m from shoreline of STP in small bay
	2	30 m from STP outfall, near south shore
	3	30 m from STP outfall, near north shore opposite warning sign of underwater cable
	4	60 m from STP outfall and 40 m from warning sign of underwater cable
	5	5 m from south shore and 60 m from STP outfall
	6	200 m from STP and 180 m from dock on south shore
	7	Adjacent to dock on south shore, 180 m from STP outfall
	8	20 m from Monument on Church Point
	9	10 m from WTP intake
	10	20 m from WTP intake
	11	10 m from island opposite WTP, 60 m from WTP
Intensive Study August 24	1	100 m from dock on shoreline, locale of STP outfall
	2	150 m from STP outfall, 50 m from south shore
	3	10 m from south shore and 150 m from STP small dock
	4	20 m from south shore at small wharf before Government Dock
	5	100 m from south shore opposite small wharf and site 4
	6	At the Government Dock, 10 m from south shore (seasonal study site 3)
	7	150 m from STP outfall opposite Church Point, north west shoreline
	8	50 m from north west shoreline next to Church Point (transect of Church Island)
	9	WTP intake, church in line with rock 250 m from shore (seasonal study site 9)
	10	200 m from WTP intake, across from WTP

STP	Sewage Treatment Plant discharge in Rossville
WTP	Water Treatment Plant

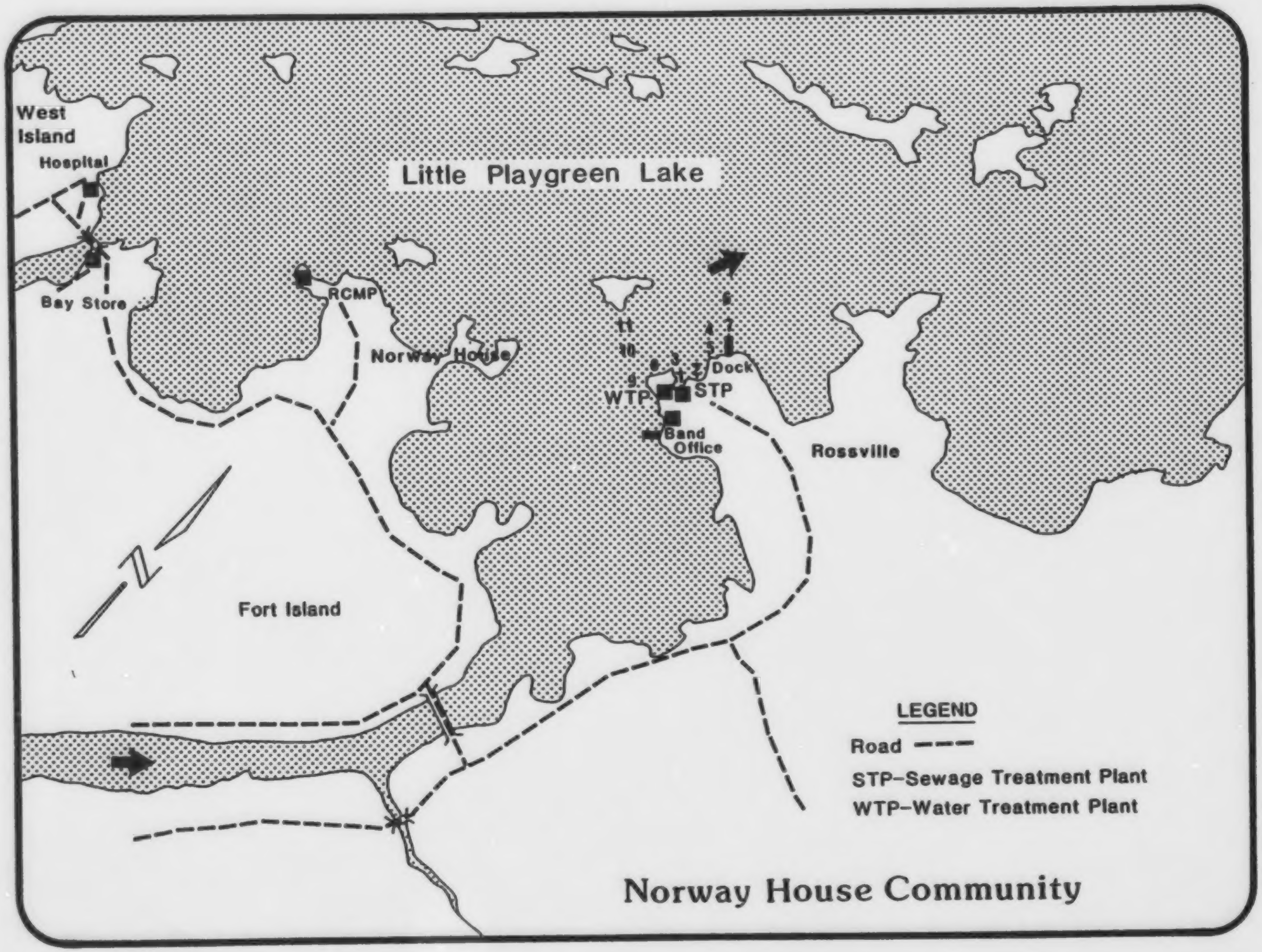

Little Playgreen Lake
West
Island
Hospital
Bay Store
RCMP
Norway House
11
10
8
9
3
1
2
4
5
6
7
Dock
WTP
STP
Band
Office
Rossville
Fort Island
N
LEGEND
Road
STP–Sewage Treatment Plant
WTP–Water Treatment Plant
Norway House Community

Figure A6. Location of bacterial sampling sites for the August 11, 1987, intensive study near the community of Norway House, Manitoba. See Table A6 for site descriptions.

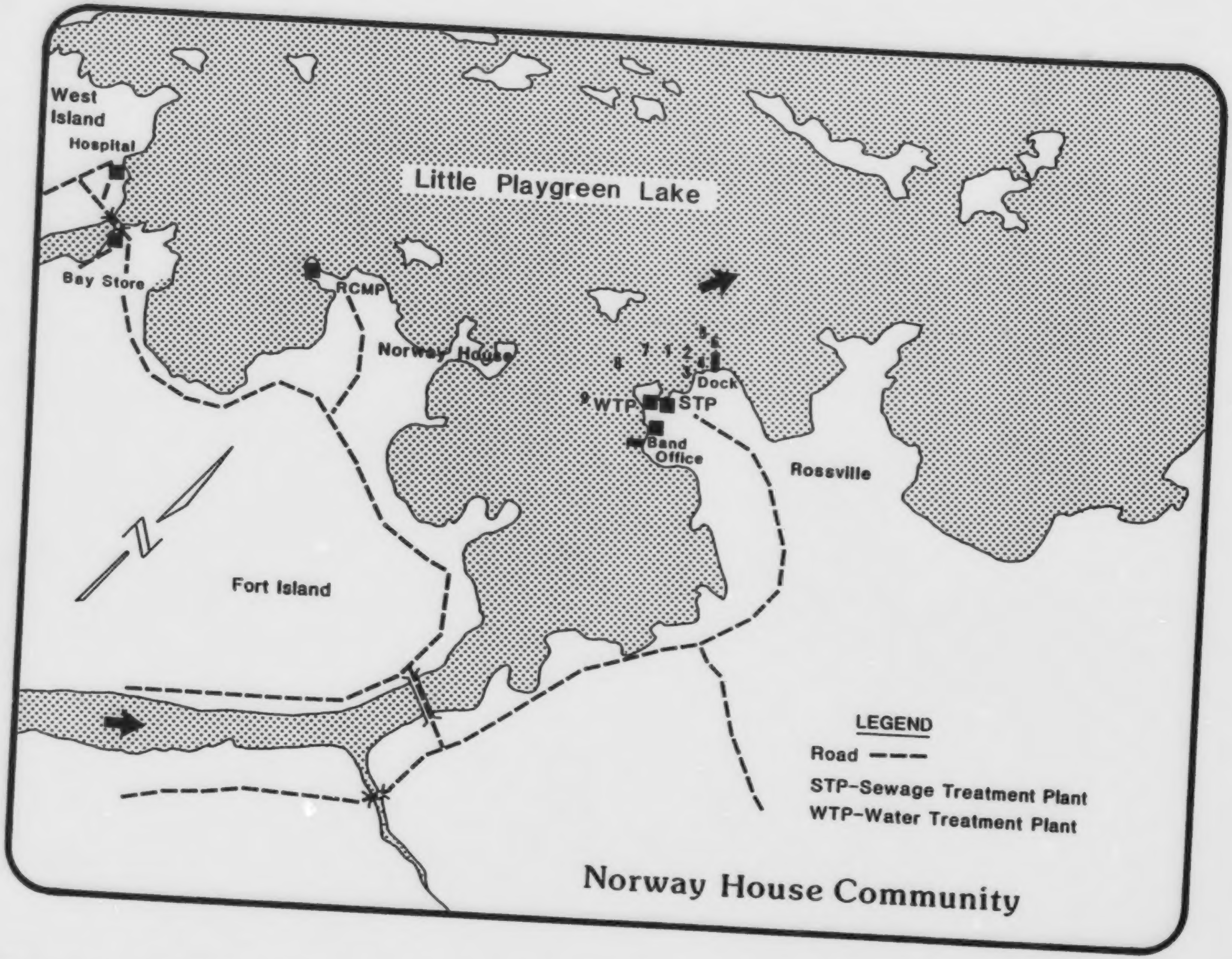

West Island
Hospital
Little Playgreen Lake
Bay Store
RCMP
Norway House
7
1
2
5
6
8
3
4
Dock
9
WTP
STP
Band Office
Rossville
Fort Island
N
LEGEND
Road ---
STP-Sewage Treatment Plant
WTP-Water Treatment Plant
Norway House Community

Figure A7. Location of bacterial sampling sites for the August 24, 1987, intensive study near the community of Norway House, Manitoba. See Table A6 for site descriptions.

Table A7. Bacteriological water quality sampling sites, Cross Lake, Manitoba, 1987. See Figures A8 and A9 for site locations.

	Site Number	Site Description
Intensive Study	1	WTP, raw water
August 11	2	150 m from wet well
	3	10 m from wet well
	4	75 m from wet well
	5	150 m from wet well, in line with intake trench
	6	10 m from wet well, in line with intake trench
	7	75 m from wet well, in line with intake trench
	8	150 m from wet well, in line with wet well and house
	9	100 m from wet well, in line with wet well and house
	10	10 m from wet well, in line with wet well and house
	11	Water Treatment Bay, at water intake, intensive study site 7
Intensive Study	1	WTP, raw water
August 25	2	10 m from wet well
	3	75 m from wet well
	4	150 m from wet well
	5	10 m from wet well, in line with intake trench
	6	75 m from wet well, in line with intake trench
	7	150 m from wet well, in line with intake trench
	8	10 m from wet well, in line with wet well and house
	9	100 m from wet well, in line with wet well and house
	10	150 m from wet well, in line with wet well and house

WTP Water Treatment Plant

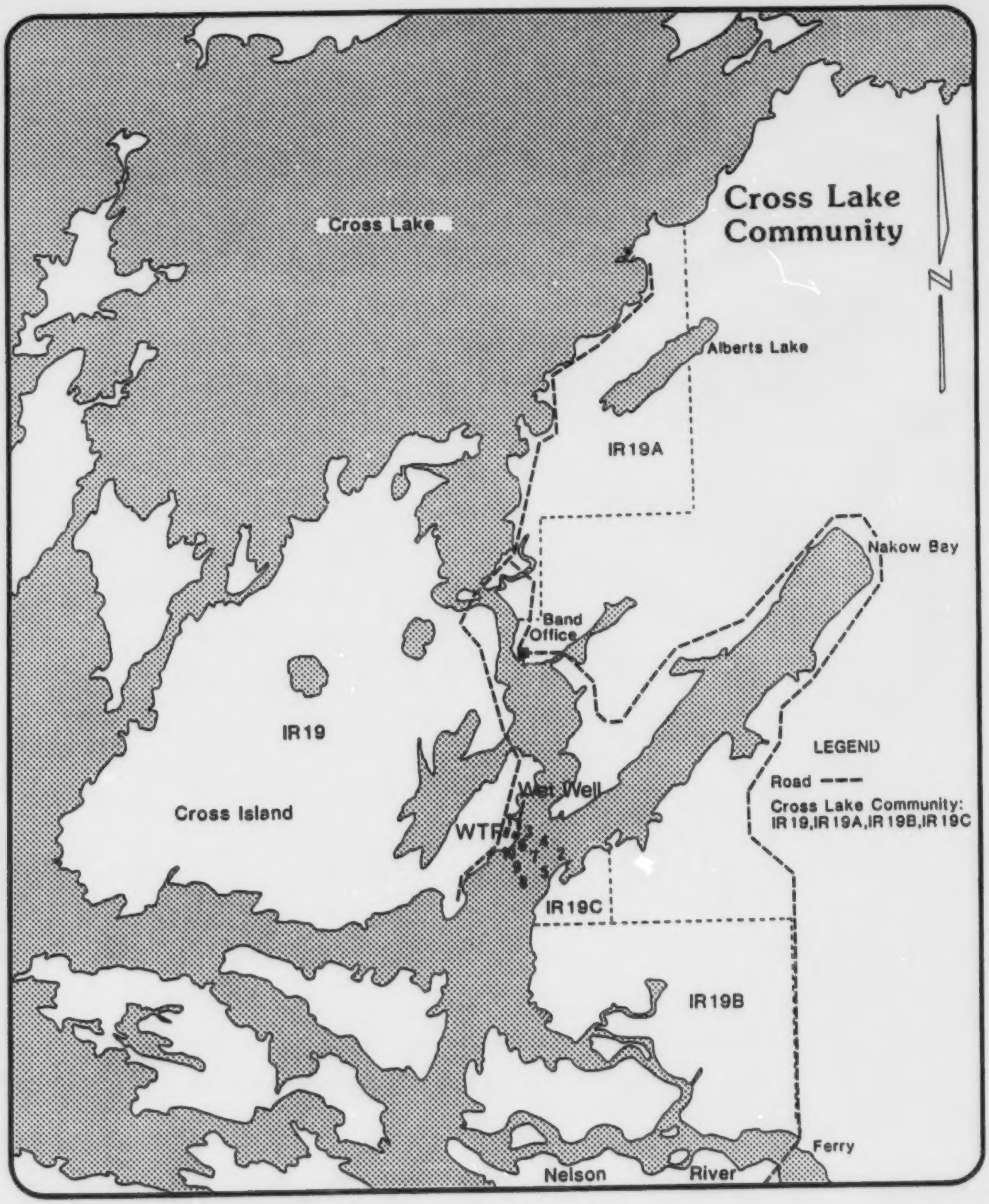

Cross Lake
Cross Lake Community
N
Alberts Lake
IR19A
Nakow Bay
Band Office
IR19
Cross Island
Wet Well
WTP
LEGEND
Road
Cross Lake Community: IR19,IR19A,IR19B,IR19C
IR19C
IR19B
Ferry
Nelson
River

Figure A8. Location of bacterial sampling sites for the August 11, 1987, intensive study near the community of Cross Lake, Manitoba. See Table A7 for site descriptions.

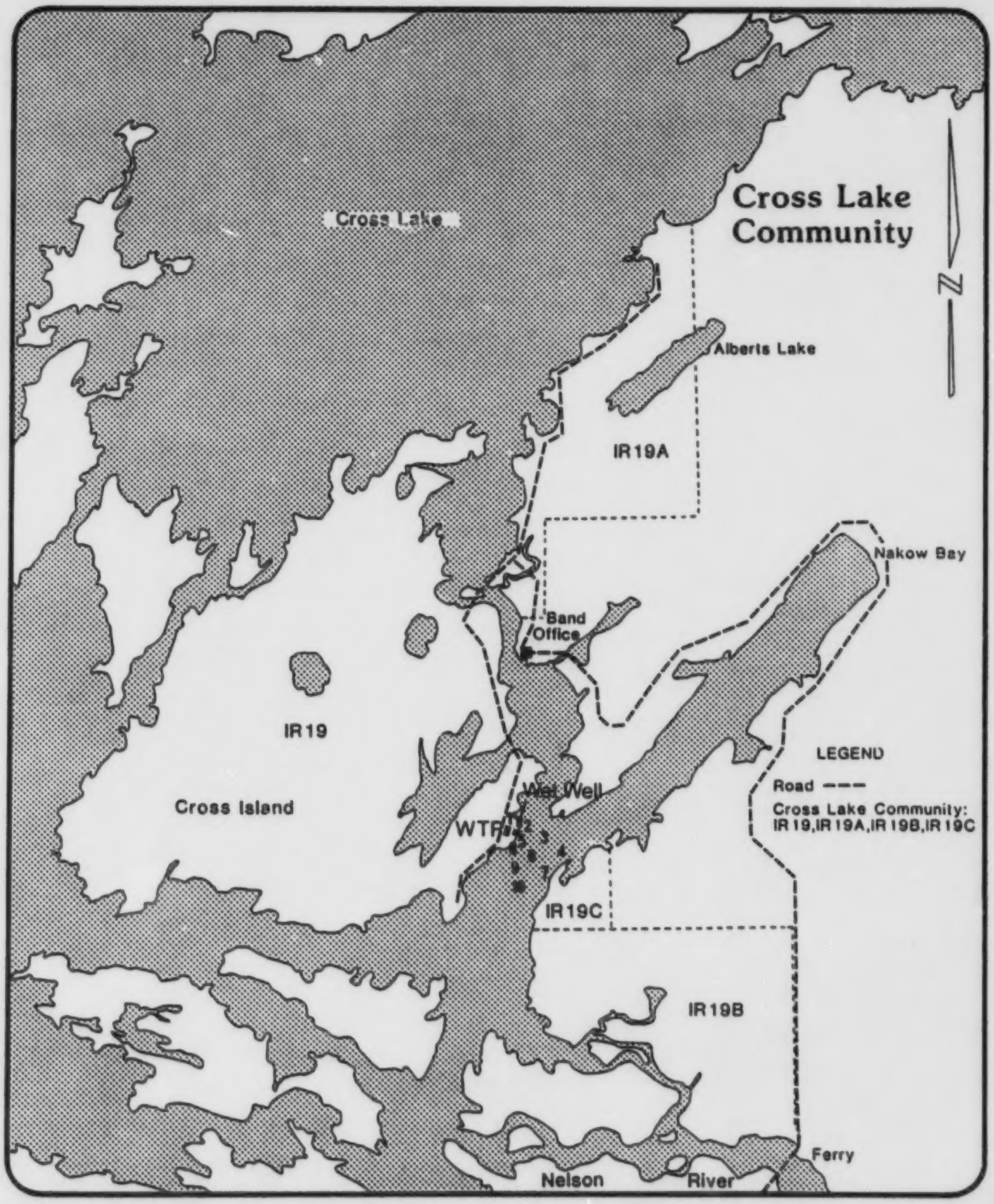

Cross Lake
Cross Lake Community
N
Alberts Lake
IR19A
Nakow Bay
Band Office
IR19
Cross Island
Wet Well
WTP
2
3
4
7
IR19C
LEGEND
Road ---
Cross Lake Community: IR19, IR19A, IR19B, IR19C
IR19B
Ferry
Nelson
River

Figure A9. Location of bacterial sampling sites for the August 25, 1987, intensive study near the community of Cross Lake, Manitoba. See Table A7 for site descriptions.

Table A8. Bacteriological water quality sampling sites, Nelson House, Manitoba, 1987. See Figures A10 and A11 for site locations.

	Site Number	Site Description
Intensive Study August 12	1	60 ft. from sewage outflow, intensive study site 6
	2	25 ft. from shore, 300 ft. from sewage outflow, intensive study site 6 (in line with lone white tree and Duncan Hall)
	3	150 ft. from shore, 300 ft. from sewage outflow, intensive study site 6, (in line with lone white tree and Duncan Hall)
	4	mid-bay, 400 ft. from sewage outflow
	5	100 ft. from shore, 400 ft. from sewage outflow
	6	40 ft. from shore, 800 ft. from sewage outflow
	7	275 ft. from shore, in line with white house on land, 800 ft. from sewage outflow
	8	850 ft. from sewage outflow, 175 ft. from shore
	9	100 ft. from shore, halfway around bend from School Bay to Water Treatment Bay (across from cemetary)
	10	120 ft. from shore, at edge of bend between School Bay and Water Treatment Bay
	11	Water Treatment Bay, at water intake, seasonal study site 7
Intensive Study August 25	1	20 m from sewer outfall in mid-bay
	2	10 m from west shore next to Duncan Hall
	3	50 m from west shore next to Duncan Hall
	4	130 m from sewer outfall in mid-bay
	5	130 m from sewer outfall 30 m from west shore (Duncan Hall)
	6	260 m from sewer outfall and 13 m from west shore next to a floating dock on point
	7	260 m from sewer outfall and 100 m from the west shore
	8	30 m from south shore, west of site 6
	9	30 m from south shore, west of site 8, next to the edge of point of land
	10	at the water intake pipe opposite the pumphouse

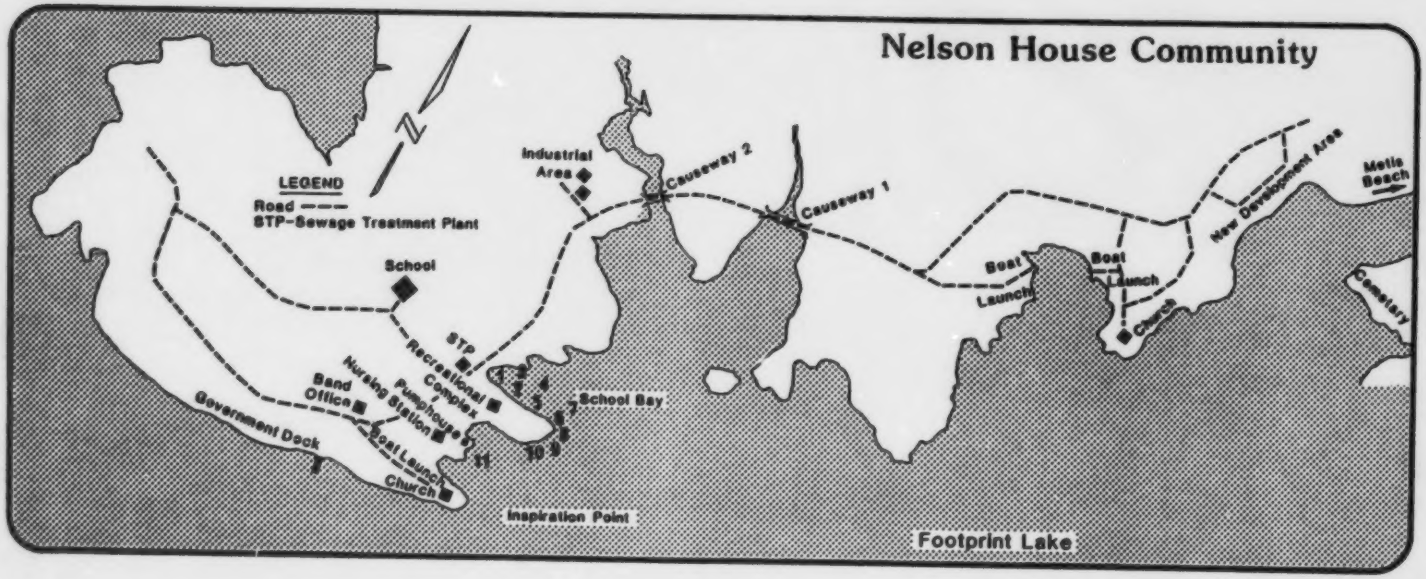
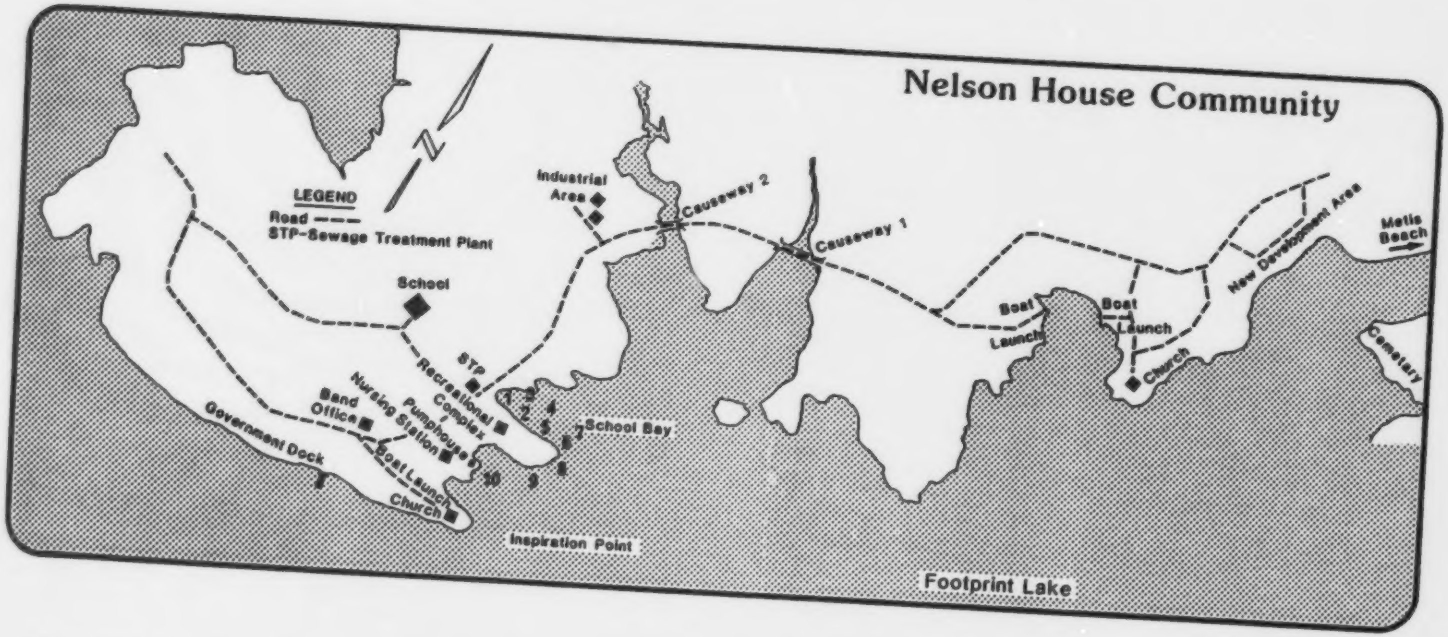

Nelson House Community
LEGEND
Road ---
STP-Sewage Treatment Plant
N
Industrial Area
Causeway 2
Causeway 1
Metis Beach
New Development Area
Boat Launch
Boat Launch
Church
Cemetary
School
STP
Recreational Complex
Pumphouse
Nursing Station
Band Office
Government Dock
Boat Launch
Church
1
2
3
4
5
6
7
8
9
10
11
School Bay
Inspiration Point
Footprint Lake

Figure A10. Location of bacterial sampling sites for the August 12, 1987, intensive study near the community of Nelson House, Manitoba. See Table A8 for site descriptions.


Nelson House Community
LEGEND
Road - - -
STP-Sewage Treatment Plant
N
School
Industrial Area
Causeway 2
Causeway 1
Boat Launch
Boat Launch
Church
New Development Area
Metis Beach
Cemetary
STP
Recreational Complex
Pumphouse
Nursing Station
Band Office
Boat Launch
Church
Government Dock
School Bay
Inspiration Point
Footprint Lake
1
2
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9
10

Figure A11. Location of bacterial sampling sites for the August 25, 1987, intensive study near the community of Nelson House, Manitoba. See Table A8 for site descriptions.

Table A9. Bacteriological water quality sampling sites, Split Lake, Manitoba, 1987. See Figures A12 and A13 for site locations.

	Site Number	Site Description
Intensive Study	1	NW corner of 1st Beach Bay past bedrock outcropping
August 12	2	NW at junction of bedrock outcrop and beach, 10 m from beach
	3	SW side of 1st beach at edge of beach and rock deposits, 10m from beach
	4	SW of STP outfall opposite large boulder on shoreline, 10 m from shore
	5	10 m from and opposite to the STP outfall
	6	30 m from and opposite to the STP outfall
	7	N side of 1st Beach Bay opposite STP outfall by 100 m and east of station 1
	8	East of station 7, 30 m from N shore of Bay
	9	S of station 8, mid-bay and 50 m from south shore
	10	10 m from S shore opposite transformer on old shack
Intensive Study	1	150 m from centroid west shore bay
August 26	2	150 m from centroid on shore of bay, between sites 1 and 3
	3	150 m from centroid east shore of bay
	4	75 m from centroid east shore of bay
	5	75 m from centroid on shore of bay, between sites 4 and 6
	6	75 m from west shore of bay
	7	20 m from centroid on mid-shore of bay
	8	20 m from east shore of bay
	9	Seasonal Study Site #12, STP in 1st Beach Bay
	10	Seasonal Study Site #11, WTP site, west of plant

STP Sewage Treatment Plant

WTP Water Treatment Plant

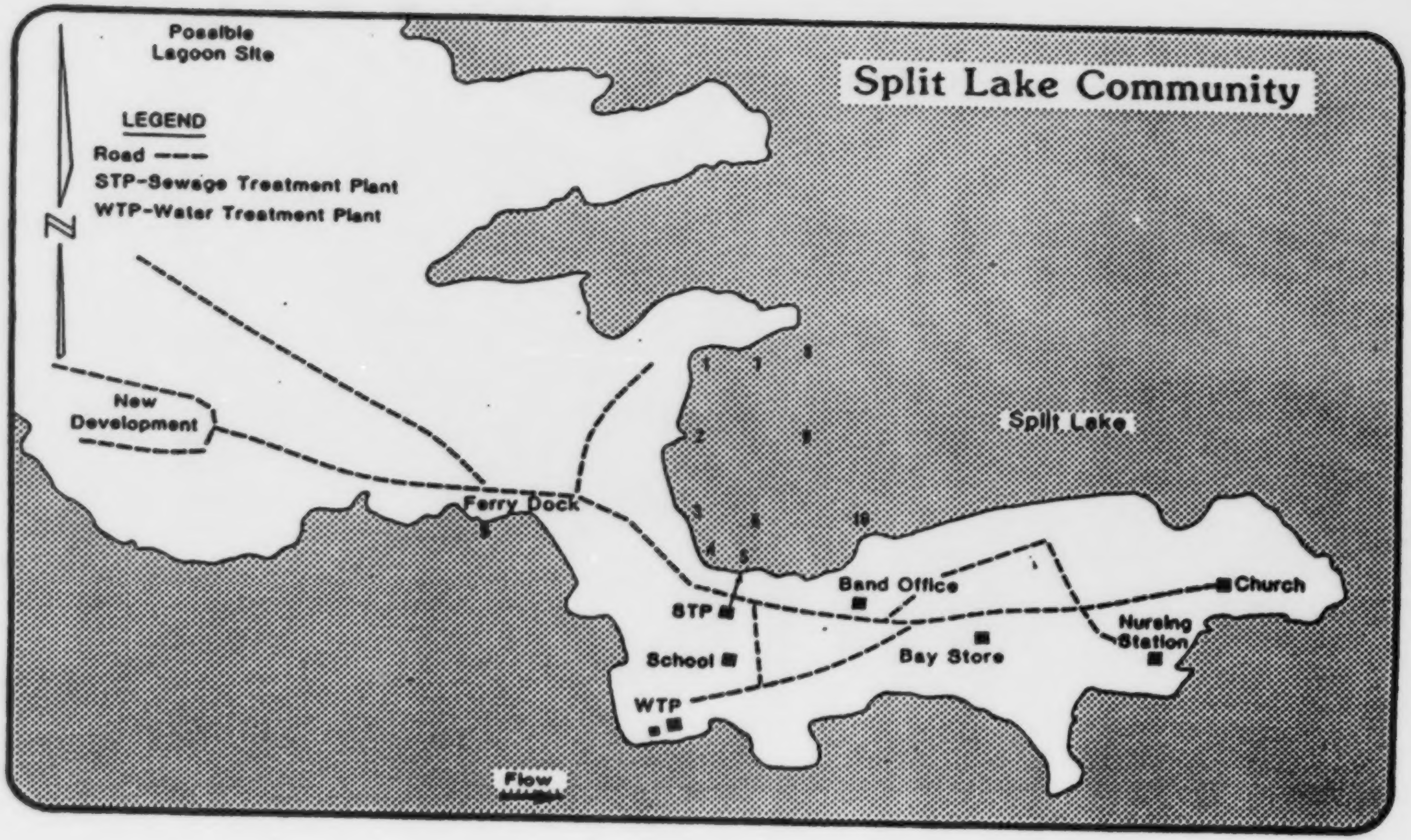
Split Lake Community
Possible Lagoon Site
LEGEND
Road
STP-Sewage Treatment Plant
WTP-Water Treatment Plant
N
New Development
Ferry Dock
Split Lake
1
7
8
2
9
3
6
10
4
5
STP
Band Office
Church
Nursing Station
School
Bay Store
WTP
Flow

Figure A12. Location of bacterial sampling sites for the August 12, 1987, intensive study near the community of Split Lake, Manitoba. See Table A9 for site descriptions.

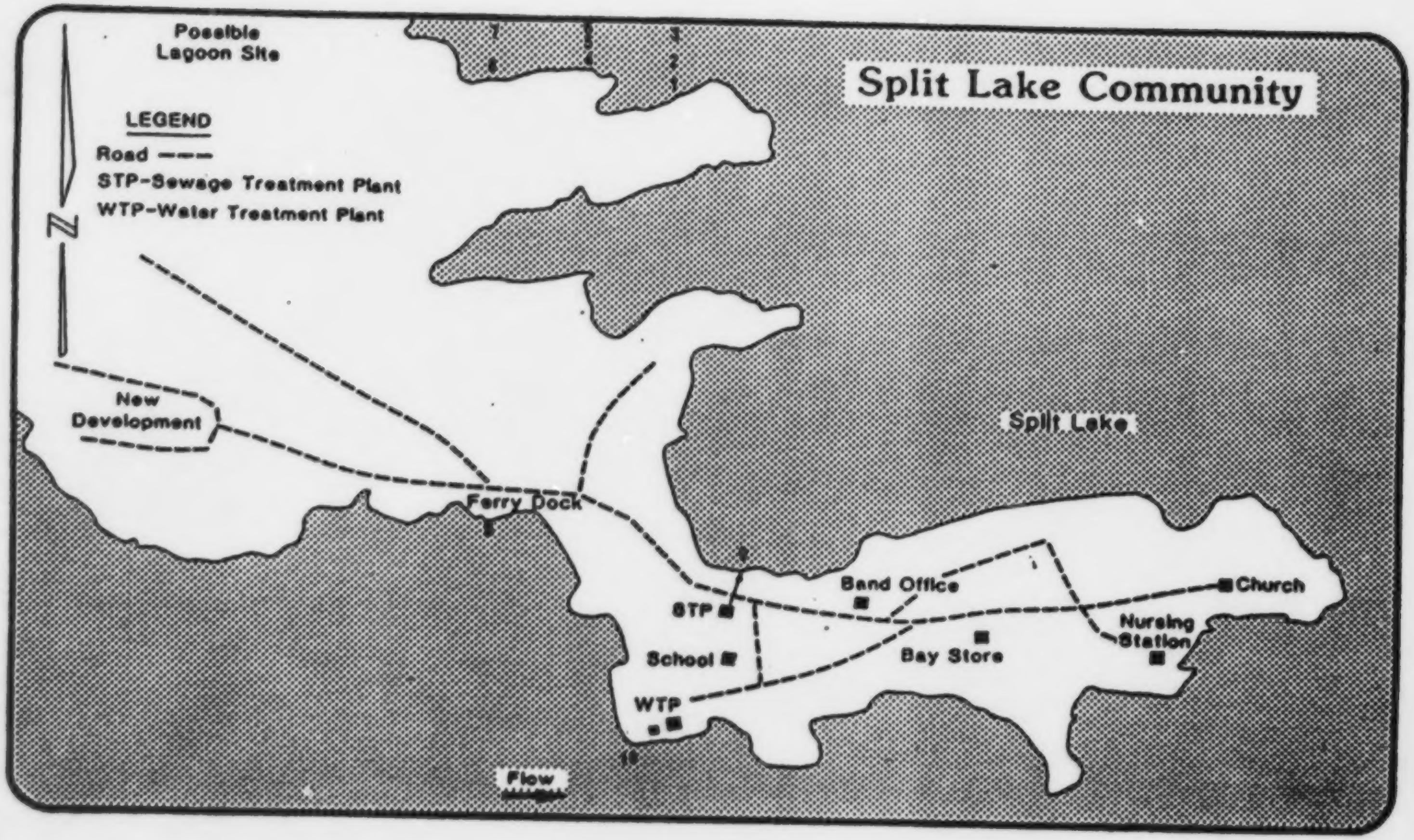
Split Lake Community
Possible Lagoon Site
LEGEND
Road ---
STP-Sewage Treatment Plant
WTP-Water Treatment Plant
N
New Development
Ferry Dock
Split Lake
Band Office
STP
School
WTP
Bay Store
Nursing Station
Church
Flow
1
2
3
4
5
6
7
8
9
10

Figure A13. Location of bacterial sampling sites for the August 26, 1987, intensive study near the community of Split Lake, Manitoba. See Table A9 for site descriptions.

THE EFFECT OF THE LAKE WINNIPEG REGULATION HYDROELECTRIC PROJECT ON THE STOCK GENETICS OF LAKE WHITEFISH (*Coregonus clupeaformis*) IN PLAYGREEN LAKE AND NORTHERN LAKE WINNIPEG: PRELIMINARY RESULTS

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ABSTRACT

The main objective of this study was to test the null hypothesis that the construction of the Lake Winnipeg Regulation Project (LWR) has not caused significant changes in the genetic relationships among lake whitefish (Coregonus clupeaformis) stocks in the LWR development area. A secondary objective was the description of the post-LWR genetic stock composition of lake whitefish in Playgreen Lake and the north basin of Lake Winnipeg. To accomplish this, the genetic composition of six spawning aggregations of lake whitefish collected in 1989 was examined and compared to the genetic composition of four spawning aggregations collected in 1975 (Kristofferson and Clayton 1990). Starch gel electrophoresis of tissue isozymes was utilized. Sixteen loci were deemed polymorphic in the 1989 sample but due to certain constraints posed by the 1975 samples, only five polymorphic loci were used for temporal analysis between 1989 and 1975. Allelic frequencies differed significantly among stocks ($P<0.05$) at two of the five polymorphic loci (MDH-B1,2) compared, indicating that lake whitefish in northern Lake Winnipeg and Playgreen Lake can be differentiated into two distinct stocks (Playgreen Lake and Lake Winnipeg). These two differentiated stocks were present before LWR and they have remained genetically distinct after the construction of LWR. These preliminary results indicate that allelic frequencies have been stable over time and support the null hypothesis that the genetic structure of the stocks has not changed since the construction of LWR. Morphological analyses should be conducted on the lake whitefish samples to determine if additional structuring exists amongst the six spawning aggregations that were sampled, and to determine if the stability seen in isozyme allele frequencies is reflected in the morphology of the spawning aggregations.

INTRODUCTION

Larkin (1972) has defined a stock of fish as a population which shares a common environment and gene pool. This definition implies a degree of reproductive isolation between various stocks of a particular fish species in a particular area. For freshwater fishes, reproductive isolation of stocks often results from the homing behavior of individual fish to return to the place of hatching. Horrall (1981) has discussed the mechanisms for the reproductive isolation of stocks of freshwater fishes. Reproductive isolation will generally be expected to result in genetic differences between stocks, which will tend to accumulate due to random genetic drift and/or selection. Larkin's definition of fish stocks also implies that different fish stocks of the same species utilize different environments for spawning, egg incubation and early life stages, when environmental influences can alter morphological traits. Therefore, differing environmental conditions will lead to morphological differences between stocks which are not based solely on genetic differences, i.e. they are at least in part environmentally induced. This is especially the case for coregonid fishes in which extreme morphological plasticity is present (Lindsey 1981).

The genetic differences expected among various stocks of fish and the environmental differences present in their habitats have allowed the development of methods to identify and differentiate stocks (Ihssen et al. 1981). Stock characteristics are usually measured in samples of fish taken from spawning areas at the time of spawning, when stocks would be expected to have segregated themselves into reproductively isolated units. Morphological differences between stocks result from genetic and/or environmental differences and these can be measured directly. Differences in behavioral or physiological characters can also be tested. The geographic limits to stock movements are often tested by the marking of fish (e.g., Pollard 1973). Direct assessment of genetic differences is possible by a number of means, the most common of which is the electrophoresis of alternate forms of functional enzymes (i.e. isozymes). Genetic characteristics, as measured by electrophoresis, are not environmentally modifiable and therefore a primary tool in identifying stocks of fish. Gel electrophoresis has been used to study intraspecific genetic variation of coregonids (Clayton and Franzin 1970; Clayton et al. 1973; Franzin and Clayton 1977; Imhof et al. 1980; Vuorinen 1984). Starch gel electrophoresis is the separation of a mixture of electrically charged molecules in an electric field through a starch gel. It is one of the most useful techniques yet devised for studying genetic variability within and among populations of organisms (Aebersold et al. 1987).

The study of fish stocks is of great importance in fisheries management and in the assessment of environmental impacts. Stocks are the natural and logical unit of management for fisheries and each stock should in theory be harvested separately because each stock can have distinct rates of growth, reproduction and mortality so fishing efforts must be distributed in accordance to the productivity of each stock. Mixed harvesting of stocks can lead to inappropriate harvest rates for individual stocks and even the extirpation of some stocks. Stock genetics is also an increasingly utilized tool in the measurement and study of environmental impacts on fish populations and fisheries. For example, Bodaly et al. (1984) were able to show that a significant change in the stock genetics of lake whitefish (Coregonus clupeaformis) had taken place in Southern Indian Lake, Manitoba and adjacent water bodies. These changes took place concurrently with disruptions in fish populations caused by the impoundment of the lake and the blockage of fish migrations via the natural outlet of the lake.

Populations of coregonid fishes in large lake systems have been shown in many instances to be divided into discrete stocks (e.g., Imhof et al. 1980; Casselman et al. 1981). In a study of lake whitefish populations in Lake Winnipeg and connecting water bodies which was based on samples collected in 1975, Kristofferson (1978 and Kristofferson and Clayton 1990), utilized morphological comparisons to identify three different stocks of lake whitefish in the north end of Lake Winnipeg and Playgreen Lake. Distinct stocks were noted for Grand Rapids on the northwest shore of Lake Winnipeg, for Big Black, Poplar and Berens Rivers on the northeast shore of Lake Winnipeg, and for Little Playgreen Lake, on the Nelson River outlet of Lake Winnipeg. Four genetic loci were found to be polymorphic but there were no significant differences in allelic frequencies for these populations (Kristofferson 1978; Kristofferson and Clayton 1990). These published results form part of the pre-development baseline for the present study.

There is evidence that there are movements of lake whitefish between Playgreen Lake and the north end of Lake Winnipeg. Some stocks migrate downstream from their feeding grounds in Lake Winnipeg to utilize Playgreen Lake for spawning during late fall and then reverse their migration upstream into Lake Winnipeg to overwinter and feed. Pollard (1973) tagged 2 934 lake whitefish at Warren Landing, at the outlet of Lake Winnipeg, between September 25th and October 7th 1970 and noted that 16 fish were recaptured downstream in the river channels (mouth of Gunisao River) by Norway House and in Little Playgreen Lake. Changes in the geographic and/or temporal distribution of flows between Lake Winnipeg and Playgreen Lake might be expected to have disrupted migration patterns between the two lakes, and possibly also the genetic structure of stocks in the area. Subsequent to the construction and operation of the Lake Winnipeg Regulation (LWR) hydroelectric project, fishermen on Playgreen Lake claimed that the spatial distributions and seasonal movements of lake whitefish in the Lake Winnipeg and Playgreen Lake areas have changed. Fishermen have also reported decreased catches of lake whitefish and have attributed this decline in catch to deleterious effects caused by the construction of LWR.

The main emphasis of this investigation was to determine whether the genetic structuring of the populations of lake whitefish in northern Lake Winnipeg and in Playgreen Lake has changed as a result of the construction and subsequent operation of LWR. This study deals with the identification of stocks of lake whitefish in northern Lake Winnipeg and in Playgreen Lake (pre-LWR and post-LWR) by utilizing biochemical genetic techniques for the analysis of genetically determined characteristics. The null hypothesis that there has been no significant change will be tested by comparing the genetic stock structure of lake whitefish populations in 1989 to the genetic stock structure present in 1975. Additional genetic analyses of 1975 samples (Kristofferson 1978) will be conducted to provide a more detailed genetic baseline. The approach deals with the testing of samples from spawning aggregations of fish for both within-group and among-group homogeneity (Shaklee 1984; Winans 1980; Richardson 1983).

The identification of post-LWR stocks of lake whitefish utilizing genetic analysis (allelic frequencies), in the Playgreen Lake and Lake Winnipeg north basin area, is a secondary objective of this study. These results should be generally useful in the management of commercial fisheries in the area.

STUDY AREA AND THE LAKE WINNIPEG REGULATION PROJECT

Lake Winnipeg is a very large (surface area greater than 24,000 km^2) lake in central Manitoba which collects water from the Saskatchewan, Red, and Winnipeg rivers and other local drainage basins. This catchment area totals more than one million square kilometers. Prior to LWR, all water exited Lake Winnipeg through Warren Landing, flowing into Playgreen Lake (Fig. 1). The majority of this flow went into the north basin of Playgreen Lake and eventually to the west channel of the Nelson River. A part of the flow entered Little Playgreen Lake and the complex of channels around the community of Norway House, becoming the East Channel downstream. The East and West channels of the Nelson joined at Cross Lake to form the Nelson River proper. The Nelson River has a mean annual flow of about 2000 m^3 s^{-1}.

The Lake Winnipeg Regulation Project was constructed by Manitoba Hydro over the period 1971-1976. The primary aim of the project was to regulate the level of Lake Winnipeg to provide larger assured winter flows into the Nelson River for hydroelectric power production on the lower Nelson. The Jenpeg control dam regulates flows through the west channel of the Nelson River, and indirectly, the level of Lake Winnipeg. The secondary aim of the LWR project was to increase the outflow capacity from Lake Winnipeg. Warren Landing is shallow and hydraulically inefficient during periods of low water levels. A variety of artificial channels and dike configurations serve to facilitate flows out of Lake Winnipeg. In the area of southern Playgreen Lake, the only change in the physical configuration of the system was the construction of Two Mile Channel (2MC) to carry a portion of the outflow of Lake Winnipeg into Playgreen Lake. The south basin of Playgreen Lake has a surface area of 275 km^2, a mean depth of about 3 m and a maximum depth of about 13 m (MacLaren Plansearch 1985). The north basin of Lake Winnipeg has a surface area of 17,520 km^2, a mean depth of 13.3 m and a maximum depth of 19 m (Brunskill et al. 1980).

Since the completion of 2MC in 1976 and the operation of LWR, the geographic and seasonal distribution of flows through the Playgreen Lake area have been altered. A higher proportion of flow now proceeds by the west channel of the Nelson River than was the case prior to LWR. Also, a significant proportion of the flow utilizing Warren Landing now leaves Lake Winnipeg via 2MC. During periods of low water levels on Lake Winnipeg, 2MC carries the majority of the flow into Playgreen Lake (e.g., almost 70% in 1977; MacLaren Plansearch 1985). During normal and high water levels in Lake Winnipeg, Warren Landing carries most of the outflow of Lake Winnipeg (about 60-70%; MacLaren Plansearch 1985). Water levels in Playgreen Lake are now lower at the end of winter (March) and higher at the commencement of winter (October) than they were before LWR, although the lake is regulated within its historic range of water levels.

METHODS

Post-LWR sampling of lake whitefish was conducted in late October 1989 at a time close to that when lake whitefish spawning was expected to occur. Samples of approximately fifty mature lake whitefish were collected by gill netting from six different sites in northern Lake Winnipeg and Playgreen Lake (Fig. 1). Thus these samples likely are representative of fish segregated into stocks. Fish were frozen and shipped to the Freshwater Institute in Winnipeg, where they were processed. For genetic (biochemical) analyses, a sample of red and white muscle and a sample of liver was removed from each fish. These tissue samples were then frozen (-30°C) for later use in electrophoretic analyses.

Frozen samples (-30°C) from four identical locations (except Gunisao R. and Two Mile Channel (Fig. 1)) sampled by Kristofferson in 1975 were obtained to determine pre-development stock structure. Biochemical analyses for these four samples were conducted using white muscle tissue extracts.

Horizontal starch gel electrophoresis was performed following the methodology outlined by Vuorinen (1984) and under the electrophoretic conditions described in Bodaly et al. (1991). Thirty-five loci were screened for samples collected in 1989 whereas 10 loci were screened for samples collected in 1975. Table 1 gives the genetic loci examined, their abbreviations, and the tissue of primary expression. Table 2 lists alleles observed at polymorphic loci with their relative mobilities on electrophoretic gels. Allele frequencies of polymorphic loci were calculated for samples from each location. Phenotypic proportions were tested for Castle-Hardy-Weinberg (C-H-W) equilibrium by Chi-square contingency tables of observed and expected phenotype frequencies. Inter-stock homogeneity was evaluated by using the G-statistic test based on maximum likelihood ratios

(Sokal and Rohlf 1981). Previous studies have shown that the following enzyme loci are likely to be polymorphic in these populations of lake whitefish and therefore useful in comparing the various spawning aggregations genetically: MDH-B1,2 (treated as two loci with equal allelic frequencies), IDDH-1,2 (SDH), G3PDH-1, G3PDH-3, LDH-B2 (liver), IDHP-3, IDHP-4, and MEP-3,4 (Bodaly et al. 1991; Casselman et al. 1981; and Kristofferson and Clayton 1990). In performing statistical analyses, BIOSYS-1 was used (Swofford and Selander 1981).

RESULTS

Genetic variation was found at sixteen of the loci screened. Eleven of these were not used for subsequent analysis: six loci had unreliable models (G3PDH-3, IDHP-3, MEP-1,2, and MEP-3,4); and only rare alleles (average frequencies over all populations <0.01) were expressed at five loci (AAT, EST, GPI-1, PGM-2, SOD). Variation found at the remaining five loci fit existing genetic models (Vuorinen 1984). Observed MDH-B1,2, G3PDH-1, IDHP-4 and LDH-B2 phenotypic frequency distributions were in compliance with expected Castle-Hardy-Weinberg distributions for both pre-LWR and post-LWR samples, with the exception that the pre-LWR Warren Landing sample G3PDH-1 allele frequencies violated equilibrium frequencies. MDH-B1,2 allele frequencies observed in 1989 differed significantly (G statistic, 3 d.f.=9.48, $P<.05$) among the subpopulations being compared (Table 3,4). Similar samples were pooled a posteriori as suggested by Kristofferson and Clayton (1990) in order to determine which stocks differed.

The samples taken in 1989 indicated that lake whitefish from Warren Landing and Little Playgreen Lake had similar MDH-B1,2 allelic frequencies (G statistic, 1 d.f.=0.32, $P>0.05$) and comprised one grouping (Fig. 2). Two additional samples from 1989 (Gunisao R. & Two Mile Channel) were grouped with similar samples from Big Black R. and Grand Rapids. Inter-group differences were significant at $P<0.05$ (G statistic, 1 d.f.=4.61).

These "pooled" groupings were similar to results obtained for the pre-development (1975) samples. Among-group allele frequency differences at the MDH-B1,2 loci for the 1975 samples were significant at $P<0.05$ (G statistic, 1 d.f.=3.80). Little Playgreen L. and Warren Landing samples were grouped together as were the Grand Rapids and Big Black R. samples (intra-group homogeneity nonsignificant at the 5% level) (Fig. 3). Allele frequencies at the MDH-B1,2 loci for the grouped samples were not significantly different between 1975 and 1989 (Warren Landing and Little Playgreen samples, 1975 vs. 1989: G statistic, 3 d.f. = 0.82, $P > 0.05$; Grand Rapids and Big Black River samples, 1975 vs. 1989: G statistic, 3 d.f. = 2.68, $P > 0.05$).

IDHP-4, G3PDH-1 and LDH-4 frequencies were not significantly different among samples at the 5% level for both pre-LWR and post-LWR sampling periods.

DISCUSSION

The analysis of MDH-B1,2 allelic frequency distributions indicated that, prior to LWR, at least two different stocks of lake whitefish existed in the area of northern Lake Winnipeg and Playgreen Lake. These two stocks were, first, fish utilizing sites in the north basin of Lake Winnipeg for spawning (Grand Rapids and Big Black River), and second, fish utilizing Little Playgreen Lake for spawning. Fish caught in the area of Warren Landing had allele frequencies similar to those caught in Little Playgreen Lake. These genetic similarities and differences were found to be present in 1975, before LWR, and in 1989, after LWR. Thus, the preliminary conclusion of this study is that there is insufficient evidence to reject the null hypothesis that there has been no significant change in the genetic stock structure of lake whitefish in the area affected by LWR. The preliminary results of this study, therefore, suggest that changes in the fishery claimed by commercial fishermen on Playgreen Lake following LWR construction and operation are not attributable to changes in stock structure of lake whitefish. This conclusion is tentative and preliminary, however, for a number of reasons. First, the analysis in this report is based on only two genetic loci. Others will probably be included in subsequent analyses as genetic models are refined and a more extensive data set is interpretable. Second, an analysis of morphological data for both 1975 and 1989 samples will eventually be included in this study. This further work may either confirm or contradict the conclusions based on the preliminary analysis presented here.

Because genetic differences indicate distinct stocks whereas similarities do not prove identity, more than two stocks may be present in the LWR area. It is in fact known already that there are more than two morphological stocks of lake whitefish in the LWR area. Kristofferson and Clayton (1990) found that the Grand Rapids, Big Black River, and Little Playgreen spawning stocks were separable on the basis of morphological characteristics. Other stocks are also probably present. Although biochemical results presented here indicated that fish captured in Playgreen Lake proper (at 2MC) and in the Gunisao River have allele frequencies at the MDH-B1,2 genetic loci similar to Lake Winnipeg stocks, this finding may be the result of coincidence. It seems more likely, based on the geographic distribution of spawning areas, that these samples represent one or two additional genetic stocks. That is, it seems unlikely that fish spawning in the Gunisao River are in fact part of the same stock as fish spawning on the west or east side of the north basin of Lake Winnipeg. One other possibility is that the 2MC stock is passing through Playgreen Lake to spawn in the Gunisao River. Further examination of biochemical genetic and morphological data will shed light on these questions.

A basic limitation of this study is the relatively small number of samples taken before LWR in the north basin of Lake Winnipeg and especially in the area of Playgreen lake. It should be emphasized, however, that the pre-LWR study was not conducted with the objective of providing a baseline data set against which to measure changes caused by LWR.

Differences in allele frequencies detected between stocks in northern Lake Winnipeg and Playgreen Lake probably reflect some degree of reproductive isolation among stocks and confer biochemical individuality upon each stock (Kristofferson and Clayton 1990). This isolation could be induced by innate homing of lake whitefish to natal spawning grounds. This would restrict gene flow between stocks and tend to perpetuate the discrete characteristics of each group. The degree of geographic isolation of stocks by distance also affects reproductive isolation by reducing effective rates of gene flow resulting from straying between adjacent spawning areas. Richardson et al. (1986) have contrasted two models which could underly the formation of genetic stocks of fishes. The isolation by distance model predicts that gradual geographically related changes occur in allele frequencies, that the effects at each locus are independent, and that there will be no geographical discontinuities in allele frequencies. The discrete stock model is supported when intra-stock allele frequencies are homogeneous and in equilibrium, when there are inter-stock differences in allelic frequencies, and when there are discontinuities in allele frequencies at the same geographical locations for several loci (Richardson et al. 1986). In the case of lake whitefish stocks, probably both distance and homing are acting to cause genetic differences between adjacent stocks.

FUTURE INVESTIGATIONS

Additional biochemical investigations combined with morphological methodologies should be carried out for the pre- and post-LWR samples. Additional stock subdivisions may be discerned. The morphological results may allow for the identification of additional morphological stocks not ascertained by genetic analysis. As mentioned above, Kristofferson and Clayton (1990) were able to discriminate three different stocks in northern Lake Winnipeg and Playgreen Lake and the possibility exists to discriminate between three more stocks by examination of post-LWR samples. Phenotypic characteristics will be of practical importance in delineating stocks and in confirming the results of biochemical analyses. In addition, Savvaitova et al. (1989) have hypothesized that environmental impacts (such as LWR) may affect the morphological heterogeneity of fish stocks. This will be tested by the application of morphological analyses in the pre- and post-LWR lake whitefish samples.

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Table 1. Enzymes screened with number of loci and tissues (M=muscle; L=liver). All loci were examined for the 1989 samples; G3PDH-1, IDHP-3, IDHP-4, LDH-B2, MDH-B1,2, MEP-1,2, MEP-3, MEP-4 only were examined for the 1975 samples.

Enzyme name	Enzyme number	Abbreviation	No. loci screened	Tissue
Aspartate aminotransferase	2.6.1.1	mAAT	1	M
		sAAT	2	M
Alcohol dehydrogenase	1.1.1.1	ADH	1	L
Creatine kinase	2.7.3.2	CK-A	2	M
Esterase	3.1.1.1	EST	1	L
Glycerol-3-phosphate dehydrogenase	1.1.1.8	G3PDH	2	M
Glucose-6-phosphate isomerase	5.3.1.9	GPI-A	2	L
		GPI-B	2	M
L-Iditol dehydrogenase [Sorbitol dehydrogenase]	1.1.1.14	IDDH	2	L
Isocitrate dehydrogenase	1.1.1.42	mIDHP	2	M
		sIDHP	2	L
Lactate dehydrogenase	1.1.1.27	LDH-A	2	M
		LDH-B	2	L
Malate dehydrogenase	1.1.1.37	sMDH-A	2	L
		sMDH-B	2	M
NADP$^+$-dependent malic enzyme	1.1.1.40	mMEP	2	M
		sMEP	2	L
Phosphogluconate dehydrogenase	1.1.1.44	PGDH	1	M,(L)
Phosphoglucomutase	5.4.2.2	PGM	2	M,(L)
Superoxide dismutase	1.15.1.1	sSOD	1	L

Table 2. Relative mobilities of alleles at polymorphic loci under electrophoretic conditions explained in the text. Alleles and locus nomenclature follow Bodaly et al. (1991) which should be consulted for allele designations in previous studies.

Locus	Allele	Mobility
G3PDH-1	a	0
	b	-85
	d	210
IDHP-3	a	100
	d	190
IDHP-4	a	100
	c	118
LDH-B2	a	100
	b	90
MDH-B1,2	a	120
	b	100
MEP-1,2	a	100
	b	40
MEP-3,4	a	100
	b	135

Table 3. Sample sizes and allele frequencies at five polymorphic loci examined in four lake whitefish samples from 1975.

Locus and alleles	Population			
	Little Playgreen	Warren Landing	Grand Rapids	Big Black River
G3PDH-1				
N	50	51	53	55
a	.250	.157	.226	.309
b	.270	.216	.245	.245
d	.480	.627	.528	.445
IDHP-4				
N	24	24	36	22
a	.896	.854	.944	.864
c	.104	.146	.056	.136
LDH-B2				
N	51	51	54	55
a	.020	.020	.028	.036
b	.980	.980	.972	.964
MDH-B1,2	50	51	53	54
a	.450	.461	.274	.380
b	.550	.539	.726	.620

Table 4. Sample sizes and allele frequencies at five polymorphic loci examined in six lake whitefish samples from 1989.

Locus and alleles	Population					
	Little Playgreen	Warren Landing	Grand Rapids	Big Black River	2MC	Gunisao River
G3PDH-1						
N	50	49	50	50	50	50
a	.220	.214	.200	.250	.210	.230
b	.320	.184	.220	.290	.300	.280
d	.460	.602	.580	.460	.490	.490
IDHP-4						
N	50	48	50	50	47	50
a	.850	.865	.790	.880	.851	.740
c	.150	.135	.210	.120	.149	.260
LDH-B2						
N	50	48	50	50	50	50
a	.040	.042	.090	.090	.100	.050
b	.960	.958	.910	.910	.900	.950
MDH-B1,2	50	48	50	50	50	50
a	.470	.510	.290	.350	.350	.370
b	.530	.490	.710	.650	.650	.630

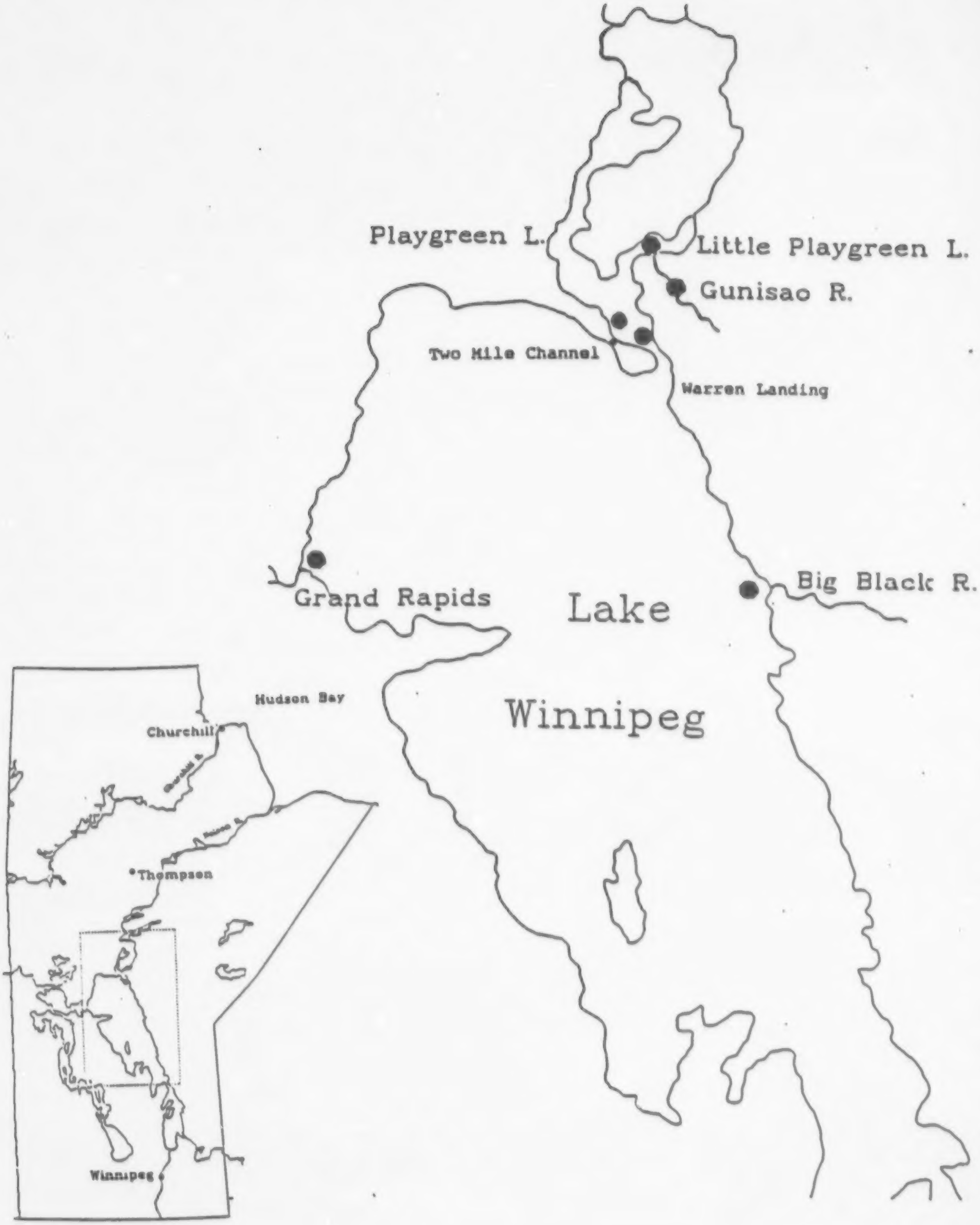
Playgreen L.
Little Playgreen L.
Gunisao R.
Two Mile Channel
Warren Landing
Grand Rapids
Big Black R.
Lake
Winnipeg
Hudson Bay
Churchill
Thompson
Winnipeg

Fig. 1. Location of sampling sites in northern Lake Winnipeg and Playgreen Lake.

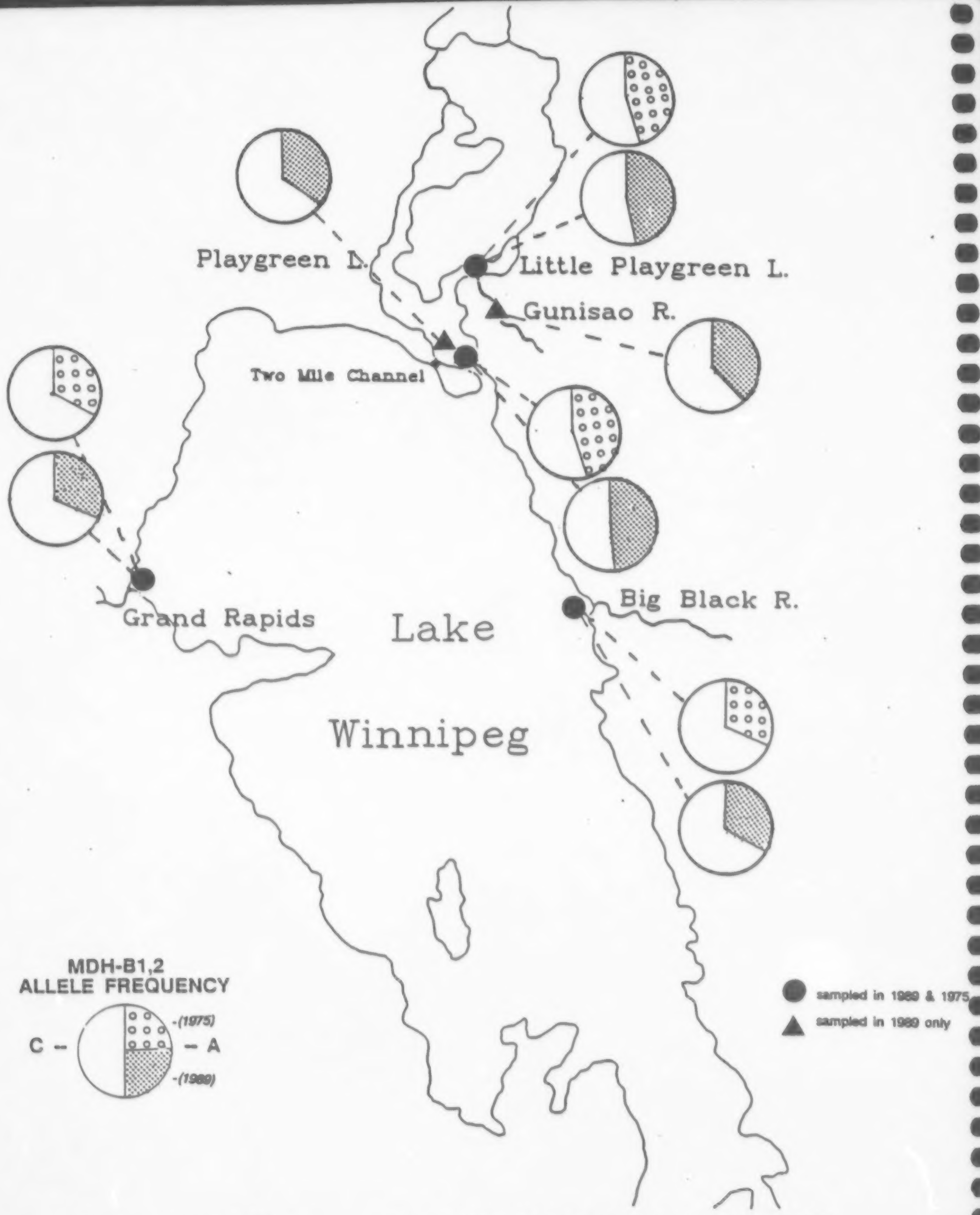
Playgreen L.
Little Playgreen L.
Gunisao R.
Two Mile Channel
Grand Rapids
Lake
Winnipeg
Big Black R.
MDH-B1,2
ALLELE FREQUENCY
C --
-(1975)
-- A
-(1989)
● sampled in 1989 & 1975
▲ sampled in 1989 only

Fig. 2. Relative allele frequencies of MDH-B1,2 alleles for post-LWR (1989) and pre-LWR (1975) samples.

Volume

2

FEDERAL ECOLOGICAL MONITORING PROGRAM

Technical Appendices

Environment Canada　Environnement Canada

Fisheries and Oceans　Pêches et Océans

FEMP TECHNICAL REPORTS

A large number of technical reports were produced under the auspices of FEMP. Most of these reports have already been published; some as part of the FEMP report series, some as part of the DFO technical report series, and some as DOE reports. In addition, there are still some reports that are in preparation. All of these reports are listed in the the following pages to provide the reader with a complete listing of FEMP sponsored studies.

The reports are listed in alphabetical order. The designation of [DOE] or [DFO] after each listing indicates the availability of each report.

For information on DOE designated reports contact

FEMP Reports
Water Resources Branch
Environment Canada
513 - 269 Main Street
Winnipeg, Manitoba R3C 1B2

For information on DFO designated reports contact

FEMP Reports
Department of Fisheries and Oceans
Freshwater Institute
501 University Crescent
Winnipeg, Manitoba R3T 2N6

The five reports published here have not been published previously.

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SPECIAL NOTE

Conclusions, recommendations or opinions expressed in reports prepared for the Federal Ecological Monitoring Program are those of the authors and do not necessarily represent those of the sponsoring departments.

Federal Ecological Monitoring Program

Final Mercury Report

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Executive Summary

This report summarizes the results of studies on mercury in the area covered by the Northern Flood Agreement conducted under the Federal Ecological Monitoring Program (FEMP) between 1986 and 1989. The primary objectives of these studies were (i) to better define the duration of the elevated fish mercury levels in flooded lakes along the Churchill River diversion and Nelson River and, (ii) to improve understanding of the factors regulating microbial mercury methylation and demethylation processes. The studies also documented mercury concentrations in fish and total and methyl mercury concentrations in water to provide information for consumers.

Concentrations of total mercury in the waters of Southern Indian Lake, and the Rat, Burntwood, and Nelson rivers have not been elevated as a result of CRD or LWR. Mean total mercury concentrations in the reservoirs range between 1.07 and 2.1 ng L^{-1} (as Hg). Although total mercury concentrations in the reservoirs are not elevated, methyl mercury concentrations in water are higher (by 55 to 245%) at reservoir sites over flooded terrain than at non-flooded reference sites. Mean methyl mercury concentrations in water sampled over reservoir sediments supporting elevated net microbial methylation ranged between 0.034 and 0.054 ng L^{-1} (as Hg) compared with 0.022 ng L^{-1} in a non-flooded reference lake. The highest concentrations occurred in the most extensively flooded reservoirs. The elevated concentrations in reservoirs were restricted to sites where net methylation in the sediments was elevated. No relationship was found between concentrations of total and methyl mercury in water. The increased methyl mercury concentrations in reservoirs of the NFA area do not represent a direct health hazard for those using these waters as a source of drinking water.

Evidence of downstream transport of methyl mercury and mercury contaminated fish, affecting fish mercury levels in waters below hydroelectric reservoirs, has been found in lakes below Notigi Dam and elsewhere in Canada. This report presents evidence which suggests flooding above Kelsey Generating Station on the Nelson River may be affecting fish mercury levels in Split Lake. Additional investigation is recommended to confirm this hypothesis.

The types of bacteria which conduct methylation and demethylation in natural lake and flooded reservoir sediments were identified using selective metabolic inhibitors. Methanogenic and sulphate reducing bacteria jointly accounted for 66% of seasonal mean methylation and 50% of mean demethylation in Granville lake sediments. The two groups were equal contributors to both processes. Conditions in the flooded zone of Methyl Bay appear to be attractive to these groups, where they accounted for all of methylation and demethylation. The proportions of total methylation and demethylation conducted by methanogens in the flooded sediments were about the same as in Granville Lake. The shares of methylation and demethylation in the flooded sediments attributable to sulphate reducers were about double those in Granville Lake.

Surveys of microbial mercury methylation and demethylation rates in sediments from a number of lakes and reservoirs representing a range of physical and chemical conditions were conducted to determine the factors which were most important in regulating net methyl mercury production. The primary limiting factor of methylation is organic matter availability, which influences methylation rates by regulating bacterial activity and by promoting development of the anoxic conditions which are favoured by the methanogenic and sulphate reducing bacteria. Methylation in sediments of lakes and reservoirs in the NFA area is not limited by inorganic mercury availability. Although methylation rates were positively correlated with total mercury concentrations in sediments, this relationship was the product of collinearity with sediment organic content.

The factor or factors which regulate microbial demethylation rates are less well understood, although organic matter and total microbial activity appear to be most important. Demethylation rates were positively correlated with both sediment organic content and total microbial activity. Sediment pH, total mercury concentration, and redox potential had no significant influence on demethylation rates although these variables often were collinear with the significant regulating factors.

An important distinguishing factor between methylation and demethylation is the lack of an influence of redox potential on demethylation rates. This may provide a partial explanation for the differential stimulation of methylation over demethylation in

flooded sediments, although additional study on the role of sediment oxygen conditions in regulating demethylation is required.

The expected duration of the elevated fish mercury levels in reservoirs of the NFA area was examined by monitoring microbial rates of mercury methylation and demethylation at flooded and reference sites in the CRD area, monitoring mercury levels in fish from flooded lakes in northern Manitoba, and measuring methylation and demethylation rates in flooded sediments from reservoirs in the boreal forest zone ranging from 24 to 59 years of age.

Six years of methylation and demethylation rate measurements in flooded sediments from Southern Indian Lake and in Granville Lake, an upstream reference lake, indicate no significant declining trend in the elevated net microbial methylation rate in flooded sediments. The differential in net methylation rate, indicated by the M/D ratio, between flooded and reference sediments was highest in 1989.

Mercury levels in whitefish, walleye, and pike from Rat and Threepoint lakes on the CRD, flooded in 1974-1976, and in Stephens Lake on the Nelson River, flooded 1970-1971, were surveyed between 1983 and 1989 by Manitoba Natural Resources, Fisheries Branch. Mercury levels have peaked in all species in all three lakes, and appear to be starting to decline. These declines are more evident in walleye and pike than in whitefish. The rate of return to pre-development mercury levels is projected to be slow, as indicated by the mean concentrations in pike and walleye from Stephens Lake which were 3 to 4 times above pre-development values 18 years after impoundment.

Mercury levels in whitefish and walleye from SIL had peaked in all regions of the lake by 1981 and had declined substantially by 1988, although not to pre-development levels. In contrast, mercury concentrations in pike had not started to decline as late as 1988, 12 years after impoundment.

The different temporal trends in whitefish and pike in SIL compared with Rat and Stephens lakes were attributed to two factors; (i) the interaction of the habitat preferences of these species with the relatively small extent of flooding on SIL compared to the other reservoirs and, (ii) the changes in limnological conditions in which have occurred in Notigi Reservoir and Stephens Lake. The declines in pike mercury levels in Rat and Stephens lakes appear related to a recent increase in shoreline erosion rates. The consequent increase in nearshore sedimentation may be reducing the area of exposed organic matter and, therefore, the area of reservoir sediments supporting elevated net methylation rates. The resulting decrease in methyl mercury availability produces lower mercury concentrations in pike. Nearshore sedimentation would not affect whitefish in these lakes because of their preference for the pelagic zone.

The greater decrease in whitefish mercury concentrations in SIL than in Rat or Stephens lakes since the initial increase following impoundment appears to be due to the offshore habits of this species and the small extent of flooding in SIL. Immediately after flooding, deposits of terrestrial organic matter in SIL extended well beyond the margin of the pre-existing lake shore as a result of offshore transport processes. While exposed, the organics would have stimulated microbial methylation in the offshore zone, contributing to mercury levels in the whitefish. The organics have since been covered by deposits of inorganic sediments, reducing offshore methylation rates to baseline levels with a consequent reduction in mercury levels in whitefish. In the much more extensively flooded Rat and Stephens lakes, whitefish are exposed to flooded sediments which support elevated net methylation rates despite their preference for the pelagic zone.

The fish mercury data for lakes on the CRD and Nelson River indicate declines are occurring in some species at some locations and these declines are proceeding in a slow and discontinuous manner. With the short data record relative to the observed rate of decline, it is difficult to base predictions of duration on the available fish mercury data. Some other approach is necessary, and measurements of the microbial methylation balance in older reservoirs appear to be the best available method.

Methylation and demethylation rates were measured in flooded sediments from reservoirs ranging from 24 to 59 years of age during the 1986-1989 period of the FEMP study in addition to annual measurements in SIL. Elevated methylation rates were found in flooded sediments from all reservoirs. Elevated demethylation rates were found in reservoirs up to 43 years of age. The elevated methylation and demethylation rates both declined over time following negative exponential decay curves. Reservoir age accounted for 64% of the variation in methylation rates and 78% of the variation in demethylation rates. Methylation and demethylation rates were projected to return to baseline 78 and 53 years, respectively, after impoundment. Net methylation, indicated by the methylation balance (M/D), declined following a second order polynomial. Extrapolation of this curve to M/D in reference lakes indicated the elevated methylation balance in SIL, lakes of the CRD, and Nelson River should return to pre-development levels 78 years after impoundment. The projected rate of decline in M/D is consistent with the observed mercury concentrations in pike and walleye from reservoirs with similar morphology and biophysical setting ranging from 12 to 47 years of age.

Factors which can affect the duration of a reservoir mercury problem include the nature of the flooded organic matter with respect to its ease of decomposition, the presence of a continuing source of fresh organic matter, and the erosion of shoreline clays. The duration of the problem in the CRD is based on the sampling of reservoirs in which boreal forest was flooded. This vegetation is resistant to decay, leading to the lengthy duration of the problem. If more readily decomposed vegetation is flooded, mercury levels in fish return to baseline much more quickly. This was the case in Cedar Lake, Manitoba, where the majority of flooded land was vegetated by marsh macrophytes and sedges and where there was no evidence of elevated fish mercury within 14 years of impoundment.

The presence of a continuing source of fresh organic matter appears to be important in maintaining fish mercury levels in

Sipiwesk Lake. The lake has frequently been subjected to low water levels in summer since construction of the Jenpeg Dam, with exposure of the drawdown zone during the growing season permitting considerable vegetation growth. This fresh organic matter is introduced to the lake during periods of high water and stimulates microbial methylation. It is expected this process will maintain the high fish mercury levels in Sipiwesk Lake for an indefinite period.

The occurrence of declines in fish mercury levels in Rat and Stephens lakes coincident with increases in shoreline erosion suggest erosion of inorganic clays may have an influence on the rate of recovery of reservoir mercury problems. Deposition of the eroded clays appears to be reducing the area of reservoir sediments supporting elevated net methylation rates. The long-term effect of this process on the rate of recovery of the mercury problem remains to be established.

The potential effectiveness of several measures for the mitigation of aquatic mercury problems or for the rehabilitation of fisheries closed due to elevated mercury levels was reviewed. Methods considered included selenium additions, increased water column turbidity, reservoir liming, inhibition of microbial methylation or stimulation of demethylation, increasing fish growth rates through intensive fishing or nutrient additions, and the selective harvest of smaller individuals. Few of these methods appear to have practical application to the reservoir mercury problem. Selenium additions can be effective in reducing mercury bioaccumulation by fish but the concentrations which must be maintained are very close to the Canadian limit for drinking water. This method is unlikely to be accepted for general application because of the high concentrations which must be maintained and, unless the effective level can be substantially reduced or it can be shown that the required level of addition is completely non-toxic, the method will be unacceptable for application to any problem. Limnocorral experiments demonstrated that increased water column turbidity is not an effective means of reducing mercury bioaccumulation by fish. Reservoir liming will be of no use in the NFA area because the elevated net methylation rates are not due to a reduction in pH. Increasing fish growth rates through nutrient additions also has no application in the NFA area because primary productivity in most of the turbid reservoirs is limited by light rather than by nutrients. The development of measures to directly manipulate the methylating and demethylating bacteria does not appear to be of any immediate use, requiring considerable additional research. In contrast, the use of intensive fishing to reduce fish mercury levels and the implementation of a size-selective commercial fishery to make use of smaller individuals of the higher value species with lower mercury concentrations both may have some limited application in the NFA area. Additional investigation is required to identify the benefits and limitations of these approaches.

The cause of the reservoir mercury problem, stimulation of microbial methylation by flooded organic matter, suggests a number of possible measures for the minimization or prevention of elevated fish mercury levels in new reservoirs. These methods include mechanical clearing of organic matter from land to be flooded, combustion of organic matter, selective removal (mechanical or combustion) of the types of organic matter which most contribute to elevated net methylation, and isolation of flooded organics from the water column by covering with inorganic soils. The only approach which can be recommended based on existing knowledge is the complete removal of all organic matter, including organic soils, prior to flooding. Further research is necessary to assess the cost, environmental impact, and effectiveness of the other approaches.

The extensive database for reservoirs in the NFA area has been employed in several attempts to develop numerical models for prediction of the severity of fish mercury problems in new reservoirs based on their physical and chemical properties. Early efforts were unsuccessful due to their failure to consider the influence of upstream reservoirs and temporal trends in fish mercury levels. Current models which incorporate these variables account for 69-84% of the variation in whitefish, walleye, and pike mercury burden in flooded lakes on the CRD. These models should be useful for prediction of the increase in fish mercury levels expected in new reservoirs planned for the CRD. Additional investigation is required to modify the models for use in other regions.

Examination of the effect of the proposed Cross Lake weir on fish mercury levels in the lake indicated that mercury levels in fish are not expected to rise following construction. The weir will raise the minimum water level without increasing high water levels above the historical maximum and no back-flooding will occur. The flooding of the re-vegetated drawdown zone of the lake also is not expected to increase mercury levels in fish. Lake levels have fluctuated considerably since construction of the Jenpeg Dam, yet mercury levels in fish have not become elevated above natural levels for the region. This appears to be due to the bedrock controlled shoreline of Cross lake which limits the re-growth of vegetation during periods of low water.

Literature on the effects of elevated mercury levels on fish physiology and behaviour was reviewed to determine if the elevated levels along the CRD represent a risk to the health of the fish. The highest muscle mercury levels recorded on the CRD since impoundment approach the upper limit of the no-effects range determined in laboratory studies. This indicates that significant effects on fish physiology and behaviour are unlikely to have occurred. Mercury levels in mink and otter from the CRD area also are generally within the no-effects range.

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Table of Contents

Executive Summary i

Acknowledgements xii

Table of Contents xiii

List of Tables xv

List of Figures xvii

1.0 Introduction 1
- 1.1 Outstanding Issues 6
- 1.2 Mercury in the Aquatic Environment 9
  - 1.2.1 Mechanisms of Microbial Methylation and Demethylation 10
  - 1.2.2 Methods of Measuring Methylation and Demethylation 12
  - 1.2.3 Relationship Between Methylation Balance and Fish Mercury 13
  - 1.2.4 Identity of Methylating and Demethylating Bacteria 16

2.0 Mercury in Water 20
- 2.1 Total Mercury 20

2.2 Methyl Mercury 26
- 2.3 Relationship Between Total and Methyl Mercury . . 28
- 2.4 Downstream Transport of Methyl Mercury 30
  - 2.4.1 Split Lake 34

3.0 Factors Regulating Methylation and Demethylation . . . 40
- 3.1 Methylation 40
  - 3.1.1 Mercury and Organic Matter 40
  - 3.1.2 Microbial Activity and Redox Potential . . 53
- 3.2 Demethylation 57

4.0 Duration of the Fish Mercury Problem 61
- 4.1 Methylation on the CRD 62
- 4.2 Mercury in Fish from SIL, the CRD, and Nelson River 64
  - 4.2.1 Rat Lake 64
  - 4.2.2 Threepoint Lake 67
  - 4.2.3 Stephens Lake 70
  - 4.2.4 Southern Indian Lake 72
  - 4.2.5 Comparison of Temporal Trends in Fish Mercury 74
- 4.3 Methylation in Older Reservoirs 81
- 4.4 Factors Affecting Duration 95
- 4.5 Duration of Mercury Problems in other Regions . . 97

5.0 Toxicity of Elevated Mercury Levels to Biota 102

6.0 Mitigation Measures 104
    6.1 Selenium Additions 104
    6.2 Increased Turbidity 105
    6.3 Increase Fish Growth 107
        6.3.1 Intensive Fishing 107
        6.3.2 Nutrient Additions 110
    6.4 Liming 111
    6.5 Inhibition of Methylation/Stimulation of Demethylation 111
    6.6 Use Fish Species or Size-Classes with Lower Mercury Levels 112

7.0 Prevention of Mercury Problems in New Reservoirs . . . 114

8.0 Prediction of Fish Mercury Levels in New Reservoirs . . 116
    8.1 Cross Lake Weir 121

9.0 Further Studies 122

10.0 References 128

11.0 Glossary of Technical Terms and Units of Measure . . . 137

List of Tables

1. Comparison of seasonal mean specific rates of microbial methylation and demethylation attributable to methanogenic and sulfate reducing bacteria; July 4 through August 24, 1988. All values are time-weighted means. Data from Ramsey (1989a). 18

2. Total and methyl mercury concentrations in water samples from lakes and reservoirs in northern Manitoba. Data for 1981 from Jackson (1988; Table 10). Data for 1987 and 1989 are from Ramsey (1988a and 1990) and are time-weighted means for the July-August period (with range). 25

3. Comparison of specific rates of microbial methylation (M), demethylation (D), and methylation balance in Graham Island, BC, stream sediments, 1988. Also listed are seasonal mean and peak M, D, and M/D in flooded forest sediments from Southern Indian Lake, Manitoba, and in Granville Lake, Manitoba, an upstream reference lake. Stream data from Ramsey (1988b). Manitoba data from Ramsey (1989a). 49

4. Chemical composition of Graham Island stream sediments; September 20-22, 1988. Listed values are the mean and standard deviation (in parenthesis) of analyses on 4 samples. Data from Ramsey (1988b). 50

5. Chemical composition of soils tested for use as substrates in the constructed wetlands and water storage reservoir. Ranges list the results of two determinations on each soil-type, otherwise values are a single determination. Data from Ramsey (1988b). 51

6. Comparison of specific rates of microbial methylation (M), demethylation (D), and methylation balance (M/D) in test-soils. Data from Ramsey (1988b). 52

7. Comparison of seasonal mean specific methylation and demethylation rates (% added Hg methylated or demethylated (g dry sediment)$^{-1}$ h^{-1}) and corresponding methylation balance (M/D) in the flooded zone of Methyl Bay and in the nearshore zone of Granville Lake; 1984-1989. Values in parenthesis are standard deviations of the mean for years when two (1984) or three (1985-1989) stations were sampled. Where no standard deviation is listed, only one station was sampled. Data from Ramsey and Ramlal (1987 a and b) and Ramsey (1987a, 1988a, 1989a, and 1990). 63

8. Total mercury concentrations ($\mu g\ g^{-1}$ wet weight) in fish from lakes on the Churchill River in northern Saskatchewan. All data from commercial fishery samples as reported by Murray (1978). The contribution of a given lake-year to the overall mean was weighted by the number of samples in that year. 94

9. Total mercury concentrations ($\mu g\ g^{-1}$) in northern pike (700 mm) in reservoirs of varying ages in Quebec and Labrador. Data from Verdon (1990). 99

List of Figures

1. Location of methylation balance study sites (▲) in northern Manitoba, 1985-1989. 5

2. Relationship between mercury uptake by caged juvenile coho salmon and (a) the specific methylation rate, and (b) the methylation balance (M/D) in stream sediments on the Queen Charlotte Islands, BC. Fish mercury data from G. Derksen, Environment Canada, Vancouver, BC. Methylation and M/D data from Ramsey (1988b). 15

3. Seasonal mean total mercury concentrations in net-plankton from lakes and reservoirs in northern Manitoba, 1984-1987. Data from Ramsey and Ramlal (1987 a and b), Ramsey (1987a), and Ramsey (1988a). 23

4. Relationship between seasonal mean total mercury concentrations in water and in net-plankton. From Ramsey (1988a). 24

5. Location of fish mercury (●) and methylation balance (▲) sampling sites on Lake Nipigon and the Little Jackfish River system, 1986. 32

6. Total mercury concentrations in spottail shiners from sites on Lake Nipigon, August 1986. Ombabika Bay 0 km was at the mouth of the Little Jackfish River. Other stations in Ombabika Bay were the indicated distance from the mouth of the river. See Figure 5 for site locations. Data from Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario. 33

7. Mean total mercury concentrations in northern pike from Split and Sipiwesk lakes, 1972-1989. Data to 1982 from Derksen and Green (1987). Later data from Green (1990). All data are from lake survey samples. 36

8. Relationship between length-standardized (350 mm) mercury concentrations in Split Lake whitefish and mean Burntwood River discharge. Discharge data (Water Survey of Canada 1990) averaged over the June to September period in each year. From Ramsey (1991). 38

9. Relationship between mean mercury concentrations in northern pike from Split and Sipiwesk lakes. Data from Figure 7. 39

10. Relationship between seasonal mean organic carbon concentrations and specific methylation rates measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data for Sokatisewin offshore station excluded from regression in (c). Data from Ramsey (1987a, 1988a, 1989a, and 1990). 42

11. Relationship between seasonal mean total mercury concentrations and specific methylation rates measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data for Sokatisewin offshore station excluded from regression in (c). Data from Ramsey (1987a, 1988a, 1989a, and 1990). 43

12. Relationship between seasonal mean organic carbon and total mercury concentrations measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data from Ramsey (1987a, 1988a, 1989a, and 1990). 44

13. Relationship between (a) seasonal mean total mercury concentrations and specific methylation rates, (b) seasonal mean organic carbon concentrations and specific methylation rates, and (c) seasonal mean organic carbon and total mercury concentrations in sediments from Methyl Bay, Southern Indian Lake, 1986. Data from Ramsey (1987a). 46

14. Relationship between (a) seasonal mean organic carbon and total mercury concentrations (b) seasonal mean organic carbon concentrations and specific methylation rates, and (c) seasonal mean total mercury concentrations and specific methylation rates in sediments from sites on the Little Jackfish River system, northwestern Ontario. Data from Ramsey (1987b). 48

15. Relationship between the seasonal mean specific methylation rate and microbial activity, as measured by the rate of dissolved inorganic carbon (DIC) production, in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. From Ramsey (1989a). 55

16. Relationship between the seasonal mean specific methylation rate and redox potential in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. Data from Ramsey (1989a). 56

17. Relationship between seasonal mean specific rates of methylation and demethylation in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data from Ramsey (1987a, 1988a, 1989a, and 1990). 58

18. Relationship between seasonal mean specific demethylation rates and organic carbon concentrations in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. Data from Ramsey (1989a). 59

19. Mean total mercury concentrations in northern pike, walleye, and whitefish from Rat Lake, 1978-1989. Pre-1983 data from Green (1986). Later data are length-standardized means from Green (1990). 66

20. Mean total mercury concentrations in northern pike, walleye, and whitefish from Threepoint Lake, 1980-1989. Pre-1983 data from Green (1986). Later data are length-standardized means from Green (1990). 68

21. Mean total mercury concentrations in northern pike, walleye, and whitefish from Stephens Lake, 1983-1989. All data are length-standardized means from Green (1990). 71

22. Location of Department of Fisheries and Oceans fish mercury sampling locations on Southern Indian Lake. From Strange et al. (1991). 73

23. Mean length-standardized mercury concentrations in whitefish from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991). 75

24. Mean length-standardized mercury concentrations in walleye from sites on Southern Indian Lake, 1975-1988 All data from Strange et al. (1991). 76

25. Mean length-standardized mercury concentrations in northern pike from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991). 77

26. Relationship between the seasonal mean specific methylation rate (% added Hg methylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b). 83

27. Relationship between the seasonal mean specific demethylation rate (% added Hg demethylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b). 84

28. Relationship between the seasonal mean methylation balance (M/D) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b). 85

29. Relationship between the elevation in seasonal mean specific methylation rate (% added Hg methylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed between 1986 and 1989. Dashed line represents equality of rates in reservoir and reference lake sediments. Granville Lake was the reference site in all years. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b, 1988a, 1989a, 1990). 86

30. Relationship between the elevation in seasonal mean specific demethylation rate (% added Hg demethylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed between 1986 and 1989. Dashed line represents equality of rates in reservoir and reference lake sediments. Granville Lake was the reference site in all years. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b, 1988a, 1989a, 1990). 87

31. Relationship between elevation in methylation balance (M/D), mean mercury concentration in northern pike, and reservoir age for sites surveyed between 1986 and 1989. The M/D decay curve was calculated from the best-fit lines in Figures 29 and 30. Pike mercury data for Stephens Lake are length-standardized (550 mm) values from Green (1990). Ogoki Reservoir data are length-standardized (550 mm) values from G. Pope, Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario. Sokatisewin Lake data are from commercial fishery samples as reported by Murray (1978). 89

32. Relationship between elevation in methylation balance (M/D), mean mercury concentration in walleye, and reservoir age for sites surveyed between 1986 and 1989. The M/D decay curve was calculated from the best-fit lines in Figures 29 and 30. Walleye mercury data for Stephens Lake are length-standardized (400 mm) values from Green (1990). Ogoki Reservoir data are length-standardized (400 mm) values from G. Pope, Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario. Sokatisewin Lake data are from commercial fishery samples as reported by Murray (1978). 90

33. Relationship between extent of flooding (% increase in lake area due to flooding) and mercury concentration in walleye in lakes on the Churchill River diversion and Nelson River, Manitoba. Fish mercury data are means of all lake survey data collected between 1980 and 1984 as reported in Bodaly et al. (1984) and by Department of Fisheries and Oceans, Inspection Services, Winnipeg, MB. Extent of flooding values for all lakes from G. McCullough (Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.). Figure modified from Ramsey (1987b) to reflect the latest (3 February 1992) extent of flooding estimates. 118

1.0 Introduction

The occurrence of elevated mercury concentrations in fish from Southern Indian Lake (SIL) and other lakes flooded by the Churchill River diversion (CRD) is a continuing source of concern for the health of the Cree and other northern populations in the area covered by the Northern Flood Agreement (NFA) which depend upon fish as a staple food resource. This mercury problem was not a predicted impact of the Churchill River diversion, yet it is among the most severe of the impacts which have been documented (Hecky et al. 1984). Consequently, mercury has been the subject of considerable study since the problem was first identified in 1979 (Bodaly and Hecky 1979); initially under the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion (CMMA) and subsequently as part of the Federal Ecological Monitoring Program (FEMP) undertaken by Environment Canada and the Department of Fisheries and Oceans (DFO), and the Ecological Monitoring Program (EMP) conducted by the Province of Manitoba. This report summarizes the findings of studies on mercury conducted by these agencies under the FEMP and the EMP, with emphasis on investigations sponsored by Environment Canada. Detailed reports of the Environment Canada studies can be found in Ramsey (1987a, 1988a, 1989, and 1990). Green (1990) and Ramsey (1991) present and analyze fish mercury data gathered under the EMP. Strange et al. (1991) present and analyze fish mercury data from SIL and Issett Lake gathered by the Department of Fisheries and Oceans.

The studies undertaken by Environment Canada concentrated on examination of the **microbial** processes which regulate **monomethyl mercury** levels in the aquatic environment. Monomethyl mercury, referred to simply as methyl mercury in this report, is important because it is the only form of mercury which accumulates in fish. **Bacteria** produce methyl mercury from inorganic mercury in the process of **methylation**. Methyl mercury is decomposed to elemental mercury and methane in the process of **demethylation** which also is conducted by bacteria. The balance between the rates of methylation and demethylation regulates the availability of methyl mercury and, in turn, determines the concentration of mercury in fish tissues. These processes are discussed in more detail in Section 1.2 of this report.

Surveys of methylation and demethylation rates in the CRD conducted under the CMMA found elevated net methylation rates in sediments taken from areas of flooded terrain but not in sediments from the pre-existing lake basins (Ramlal et al. 1987; Ramsey and Ramlal 1987 a and b). Rates of both methylation and demethylation were elevated in flooded sediments, but methylation was differentially stimulated resulting in the elevated net methylation rate. **Limnocorral** experiments indicated the most important factors regulating net mercury methylation were the amount and type of flooded organic matter (Hecky et al. 1987a).

Early studies of fish mercury levels in reservoirs located in the central and southern United States suggested the elevated levels were transient and would return to baseline levels 3 to 5 years after impoundment (Cox et al. 1979; Abernathy and Cumbie 1977). The CMMA studies found this estimate was not applicable to northern reservoirs. Based on the observation of elevated net methylation in the flooded zone of a 40 year old reservoir and the lack of a significant downward trend in mercury levels in predatory fish species from the CRD, it was speculated that several decades would be required for recovery (Hecky et al. 1987b). A more precise estimate of the duration was not possible, however, due to the lack of information from older reservoirs in northern Canada.

The CMMA studies also found there was no proven means of mitigating reservoir mercury problems. It was suggested that an area of investigation which might be followed was the manipulation of the methylating or demethylating bacteria in reservoirs to reduce net methyl mercury production. This approach required greater knowledge of the causative micro-organisms and their limiting factors than was available at the start of the FEMP. Thus, the primary goals of the FEMP mercury program were:

i). to better define the duration of the mercury problems in flooded lakes along the Churchill River diversion and Nelson River, and;

ii). to provide a better understanding of the factors regulating methylation and demethylation to determine if it is possible to regulate methyl mercury production in reservoirs.

Two approaches were used in assessing duration. Methylation and demethylation rates were measured annually, from 1986 to 1989, at a site on SIL and in Granville Lake (Figure 1), a non-flooded reference lake, to look for evidence of a decline in the methylation balance in flooded sediments over the short term. The SIL site was located in Methyl Bay, a small protected bay near the community of South Indian Lake (Figure 1), and included stations in flooded terrain and in the pre-existing lake basin. The same measurements were made in sediments from the flooded zones of a number of older reservoirs in Manitoba, Saskatchewan, (Figure 1) and northwestern Ontario, ranging from 24 to 59 years of age, to examine long-term trends in methylation and demethylation activity. These sites represented a broad range of physical and chemical conditions, enabling examination of the significance of various environmental factors in the regulation of microbial methylation and demethylation. The groups of bacteria responsible for methylation and demethylation in SIL and Granville Lake also were identified using specific metabolic inhibitors.

Studies of mercury on the Churchill River diversion and Nelson River have largely concentrated on measurements of mercury levels in fish, and on examination of the processes involved in transformations of mercury in the aquatic environment. These studies were designed to serve the information requirements of the various government agencies with responsibilities in the area and to satisfy the perceived information needs of the affected communities. However, it is possible that some issues of importance to

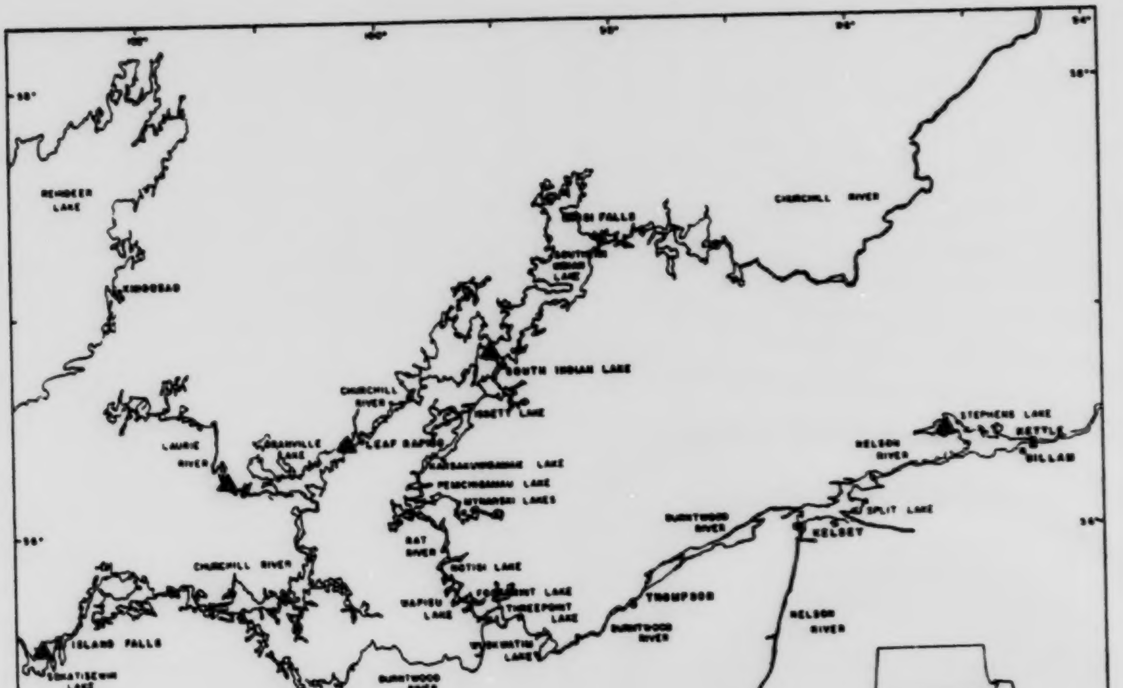

102°
100°
98°
96°
94°
58°
56°
54°
REINDEER LAKE
KINOOSAO
MISSI FALLS
SOUTHERN INDIAN LAKE
CHURCHILL RIVER
SOUTH INDIAN LAKE
CHURCHILL RIVER
ISSETT LAKE
LAURIE RIVER
GRANVILLE LAKE
LEAF RAPIDS
KARSAKUWIGAMAK LAKE
PEMICHIGAMAU LAKE
MYNARSKI LAKES
RAT RIVER
CHURCHILL RIVER
NOTIGI LAKE
FOOTPRINT LAKE
WAPISU LAKE
THREEPOINT LAKE
WUSKWATIM LAKE
ISLAND FALLS
SOKATISEWIN LAKE
BURNTWOOD RIVER
STEPHENS LAKE
KETTLE
NELSON RIVER
GILLAM
SPLIT LAKE
BURNTWOOD RIVER
KELSEY
THOMPSON
NELSON RIVER
BURNTWOOD RIVER
SIPIWESK LAKE
CROSS LAKE
JENPEG
NELSON RIVER
MANITOBA
MOOSE LAKE
SASKATCHEWAN RIVER
CEDAR LAKE
CROSS BAY
GRAND RAPIDS
LAKE WINNIPEG
N
LEGEND
GENERATING STATION
CONTROL STRUCTURE
0 20 40 60 80

Figure 1. Location of methylation balance study sites (▲) in northern Manitoba, 1985-1989.

specific communities remain to be identified and examined. In this regard, a survey of community opinion with respect to outstanding mercury-related issues was undertaken. This survey was by no means exhaustive, as it was not possible to directly solicit public views through meetings, workshops, or community surveys. Instead, people in the communities with a good understanding of the community information requirements, and a position in which they would become aware of mercury-related issues through their daily activities, were interviewed. These people generally were the communities' key communicators. Information from the survey of communities was augmented with information from interviews with representatives of government agencies, consultants working in the area, and Manitoba Hydro. The survey was restricted to non-medical issues.

1.1 Outstanding Issues

One issue of particular importance identified by several of the people contacted is the problem of translating the word mercury into the Cree language. In the CMMA information program, mercury was interpreted as "something like a poison, in that too much is harmful" (L. McKerness, Environment Canada, Winnipeg, MB. pers. comm.). This definition is problematic in that the word poison seems to have become the focus of attention. There is a general anxiety regarding mercury levels in fish throughout the NFA area, including the Cross Lake community where mercury levels in fish are low. It has since been suggested that the term "fish sickness" be used. This also is an imprecise translation because it does not convey the relationship between the amount of mercury in a fish and the relative hazard to health and it implies that the fish become sick as a result of high mercury levels, which does not seem to be the case (Section 5.0). Any attempt to communicate the conclusions of this and other mercury studies in Cree will be limited by the language barrier yet there is the risk that an incomplete translation may cause unnecessary anxiety. In the absence of a satisfactory translation, it has been suggested that the english word, mercury, be used. This approach is not entirely satisfactory but may help to prevent misunderstanding related to imprecise translation.

There also seems to be a need for the presentation of information on the mercury issue in a manner that is understandable to people without a technical or scientific background. The communication of information on mercury and other issues to the NFA communities has been a high priority of the FEMP. However, my discussions with the communities indicate the program has not been entirely successful. More interactive approaches appear to be necessary to promote fuller understanding of the reservoir mercury problem. These would require active community participation, perhaps in the form of workshops dealing with specific issues.

The question of the effect of the elevated mercury levels in fish and other animals on the health and behaviour of the animals also was raised. Only one of the studies conducted under the CMMA mentioned toxicity to wildlife (Kucera 1987), and none of the studies conducted thus far has examined the potential significance of mercury-related health effects on fish and wildlife in any detail. Section 5.0 in this report summarizes current knowledge on the effect of elevated mercury levels on the behaviour and health of fish and aquatic mammals.

Among the major outstanding questions with regard to mercury in the Split Lake area is whether the upstream hydroelectric developments on the Nelson River and Churchill River diversion have affected in the past, or are continuing to affect, mercury levels in fish from the lake (A. Beardy, Split Lake, MB, and W. Wysocki, Symbion Consultants, Winnipeg, MB, pers. comm.). This question is considered in Section 2.4.1.

Another concern of Split Lake residents is the lack of information on mercury levels in biota other than fish. The "country" food base of the community includes many products of fish-eating birds (e.g., gull eggs and mergansers) and mammals (e.g., bear) for which no information has been published.

The proposed weir near the outlet of Cross Lake for the maintenance of water levels also has raised the question of possible effects on mercury levels in fish (L. McKerness, Environment Canada, Winnipeg, MB, pers. comm.). This concern would appear to be a product of experiences in other northern communities where mercury levels in fish have increased after a rise in water levels. The potential effect of this weir on fish mercury levels in Cross Lake is evaluated in Section 8.1 of this report.

1.2 Mercury in the Aquatic Environment

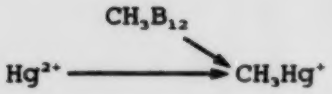
Mercury is a naturally occurring metal and can be found, in varying levels, in all soils, water, and biota. The ultimate source of all mercury in the environment is the inorganic mercury in bedrock (Andersson 1979). Although originally derived from this source, the mercury found in fish occurs in an organic form called monomethyl mercury. Long considered to be the dominant form of mercury in fish muscle tissue (Huckabee et al. 1979), recent research has shown that methyl mercury is the only form which accumulates in fish muscle (Bloom 1989). Any other forms which have previously been reported as occurring in fish muscle are the product of sample contamination in the field or laboratory.

In aquatic systems, methylation of inorganic mercury by bacteria is the only important means of methyl mercury production. Abiotic, or chemical, methylation also has been shown to occur, but is thought to account for less than 10% of total methyl mercury production (Jensen and Jernelov 1969; Furutani and Rudd 1980; Berman and Bartha 1986). Once produced, methyl mercury is very stable in the natural environment. The only known abiotic mechanism of methyl mercury decomposition is photolysis. However, the low sunlight absorption rate constant for the methyl mercuric ion (CH_3Hg^+) is thought to preclude photodecomposition as a significant degradation pathway (Baughman et al. 1973). The only other known natural means of decomposition is demethylation by bacteria (Bisogni 1979).

Methyl mercury is of particular concern for human health because of its ability to cross the blood-brain barrier and concentrate in the central nervous system (Carty and Malone 1979), causing a variety of nervous disorders. Methyl mercury also has a strong tendency to accumulate in higher animals, such as fish, furbearers, and humans, because of its strong affinity for **sulfhydryl groups** in **proteins** (Carty and Malone 1979). The report by Health and Welfare Canada (1987) can provide additional information on the effects of methyl mercury on human health and the levels which place a person at risk of health impairment.

1.2.1 Mechanisms of Microbial Methylation and Demethylation

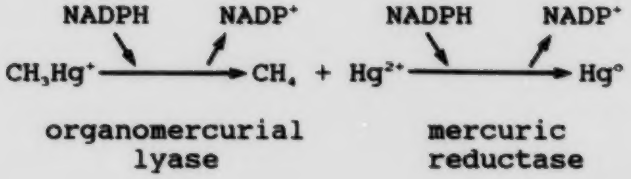
The mechanism of biological methylation remains to be fully described but, at present, is thought to involve the **nonenzymatic** transfer of methyl groups (CH_3^-) from **methylcobalamin (Vitamin B_{12})** to **Hg^{2+} (mercuric ions)** in bacterial cells (DeSimone et al. 1973; Robinson and Tuovinen 1984). The three major **co-enzymes** involved in methyl transfer reactions in cells are N5-methyltetrahydrofolate derivatives, S-adenosylmethionine, and methylcobalamin. Methylcobalamin is considered to be the co-enzyme responsible for methylation of Hg^{2+} salts because it is the only one capable of transferring **carbanion** methyl groups (Bertilsson and Neujahr 1971; DeSimone et al. 1973). The reaction proceeds by the **electrophilic** attack of the mercuric ion (Hg^{2+}) on the methyl (CH_3) group of Vitamin B_{12} to produce the methyl mercuric ion (CH_3Hg^+) as follows:



$$Hg^{2+} \xrightarrow{CH_3B_{12}} CH_3Hg^{+}$$

The occurrence of Vitamin B_{12} in most bacteria and the lack of need for a specific enzyme to facilitate transfer of the methyl group means that many different types of bacteria are capable of mercury methylation. It is not a specialized process, rather it can be viewed as an accident of normal cell metabolism. Although non-enzymatic methylation appears to be the most common mechanism, enzymatic transfer of methyl groups (CH_3^-) to mercuric ions (Hg^{2+}) also has been suggested to occur as some bacteria lacking methylcobalamin have been shown to methylate mercury (Landner 1971).

Demethylation is the better understood of the mechanisms regulating methyl mercury availability. The process has two enzyme-mediated steps: (1) the cleavage of the carbon-mercury bond of the methyl mercuric ion (CH_3Hg^+) by **organomercurial lyase** to produce methane (CH_4) and a mercuric ion (Hg^{2+}); and, (2) the reduction of Hg^{2+} to elemental mercury (Hg^o) by **mercuric reductase** (Robinson and Tuovinen 1984; Summers and Silver 1978) as follows:



$$CH_3Hg^{+} \xrightarrow[\text{organomercurial lyase}]{NADPH \searrow \quad \nearrow NADP^{+}} CH_4 + Hg^{2+} \xrightarrow[\text{mercuric reductase}]{NADPH \searrow \quad \nearrow NADP^{+}} Hg^{o}$$

The requirement for two specialized enzymes makes the ability to demethylate a trait found only in those bacteria which carry the genetic information required to produce these enzymes. The genetic information for production of organomercurial lyase is encoded on transposons (sexually transmitted plasmids; Pan-Hou et al. 1980) rather than on the bacterial chromosome, so that the number of plasmids does not necessarily increase with the size of the microbial community. This is one possible explanation for the differential stimulation of methylation over demethylation in reservoir sediments (Ramsey 1987a).

1.2.2 Methods of Measuring Methylation and Demethylation

The specific rate of mercury methylation was determined in the FEMP studies using the **radioisotope** method developed by Furutani and Rudd (1980) which measures the rate of methylation of $^{203}Hg^{2+}$. The specific rate of demethylation was determined using the radioisotope method of Ramlal et al. (1986) which measures the rate of decomposition of $^{14}CH_3HgI$. Unlike other methods of measuring rates of mercury transformations, both methods used here minimize sample incubation time and the quantity of mercury which must be added so that the microbial community is not altered from that in situ, as indicated by linear methylation and demethylation kinetics over the incubation period (Furutani and Rudd 1980; Ramlal et al. 1986).

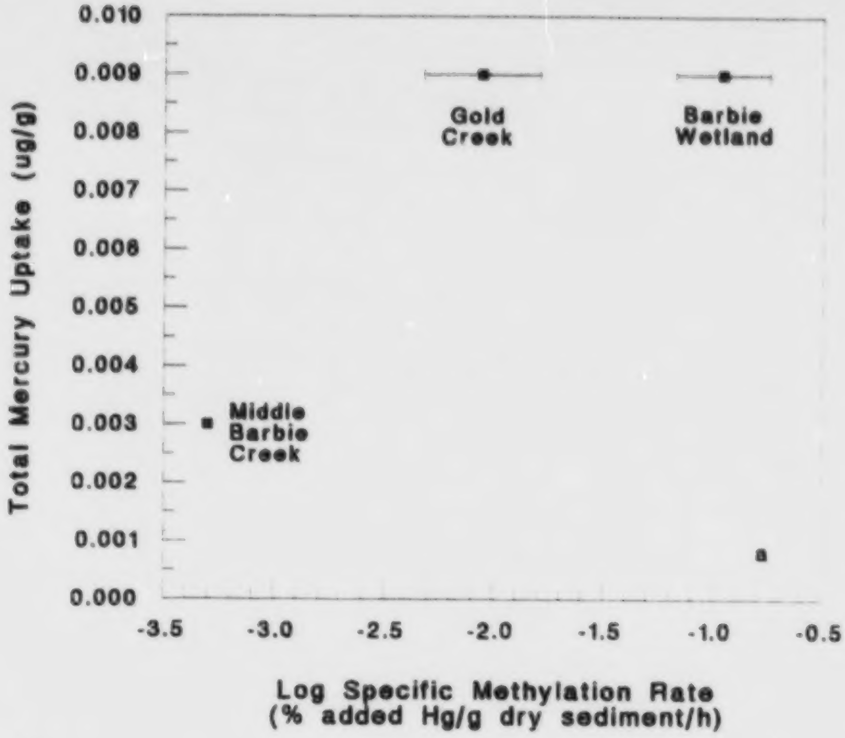
The rates derived with each method are termed specific because both processes are dependent upon mercury concentration and, as some mercury is added to the samples, the rates measured are specific to the quantity added. Addition of the same quantity of mercury to all samples should make differences in specific rates between locations proportional to differences in the **in situ** rates (Furutani and Rudd 1980). If concentrations of the bioavailable mercury species (Hg^{2+} for methylation, and CH_3Hg^+ for demethylation) were known, the specific rates could be converted to absolute rates. However, analytical methods for these mercury species in sediments have yet to be developed, restricting expression of the rates to the specific form. The only practical limitation of leaving the measured rates in the specific form is that the difference between the specific methylation and demethylation rates does not yield an absolute net methylation rate. Instead, the ratio of these rates (**M/D**), or what we call the **methylation balance**, is used as an indicator of the relative difference in the net rate of methyl mercury production between locations.

1.2.3 Relationship Between Methylation Balance and Fish Mercury

Traditional methods of measuring mercury methylation rates employ additions of inorganic mercury, incubation periods on the order of several days to weeks, and measurements of methyl mercury yield by gas chromatography (e.g., Jackson 1987). These methods are considered to measure net methylation rates because the amount of methyl mercury measured at the end of the incubation period represents what is left over after the processes of methylation and demethylation have acted. It might be reasonable to assume the same is true for the radiochemical method used to measure specific methylation rates in the FEMP studies. This does not seem to be the case, however, as indicated by an experiment conducted in streams on the Queen Charlotte Islands, British Columbia (Ramsey 1988b).

The experiment involved the use of caged juvenile coho salmon (*Oncorhynchus kisutch* Walbaum) to monitor methyl mercury availability at three locations; Gold Creek, Barbie Wetland, and Middle Barbie Creek. Measurements of microbial methylation and demethylation were made at the same sites using the radioisotope methods described above.

Mercury uptake by the fish was the same at two of the sites, Gold Creek and Barbie Wetland (Figure 2). Uptake at these sites was significantly higher than at Middle Barbie Creek. The specific methylation rates in Gold Creek and Barbie Wetland sediments also were significantly higher than at the Middle Barbie Creek site, but the specific methylation rate in Barbie Wetland was significantly higher than in Gold Creek (Figure 2). M/D in Gold Creek and Barbie Wetland was significantly higher in Middle Barbie Creek and there was no significant difference in M/D between Gold Creek and Barbie Wetland. Methyl mercury uptake by the fish was proportionate to the methylation balance (M/D) but not to the specific methylation rate (Figure 2). This indicates M/D is a measure of the net methylation rate while the specific methylation rate is not, contrary to traditional analytical methods. All studies of microbial methylation and demethylation conducted under the CMMA and FEMP which have used the radiochemical analytical methods have employed



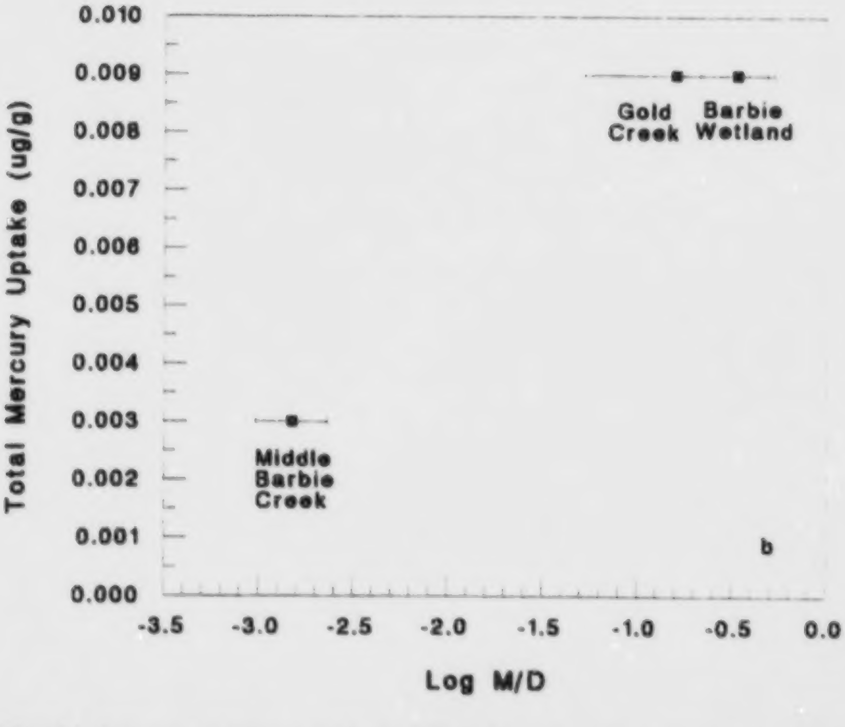


Figure 2. Relationship between mercury uptake by caged juvenile coho salmon and (a) the specific methylation rate, and (b) the methylation balance (M/D) in stream sediments on the Queen Charlotte Islands, British Columbia. Fish mercury data from G. Derksen, Environment Canada, Vancouver, BC. Methylation and M/D data from Ramsey (1988b).

M/D as a measure of the net methylation rate.

For a methylation rate measured using the radiochemical method to be a net value, **isotopic equilibrium** between the radio-labelled methyl mercury produced during the incubation and the non-labelled methyl mercury, both present in the sediments at the start of the assay and produced during the assay, must be attained early in the incubation period. With a 12 hour incubation period, and very slow methylation rates (between 0.0005% and 0.03% added Hg (g dry sediment)$^{-1}$ h^{-1}), it is unlikely that equilibrium is approached during the assay. The radiochemical method probably does not yield a true gross methylation rate, as some of the radio-labelled methyl mercury produced in the assay undoubtedly is demethylated before the end of the incubation. However, the failure of the methylation rate alone to account for mercury uptake by the caged fish indicates the rate is much closer to the gross than to the net value.

1.2.4 Identity of Methylating and Demethylating Bacteria

The methylating and demethylating bacteria were identified, at the level of major metabolic group, using selective metabolic inhibitors. These inhibitors prevent a specific type of bacteria (e.g., **methanogenic**) from conducting the chemical reactions necessary for growth and reproduction without substantially affecting the activity of other types of bacteria. Two groups of bacteria, **methanogens** and **sulphate reducers**, were singled out for study as other investigations had shown they were the most likely to be important in mercury transformations (Wood et al. 1968; Compeau and Bartha 1985). Bromoethane sulfonic acid (BESA), a specific inhibitor of **methanogenesis** (Gunsalus et al. 1978), was used to examine the role of methanogenic bacteria in methylation and demethylation. Sodium molybdate, a specific inhibitor of sulphate reduction (Postgate 1949; Saleh et al. 1964), was employed to identify the importance of sulphate reducing bacteria in these processes. The importance of each group was determined by measuring the reduction in the specific methylation and demethylation rates caused by the addition of each inhibitor. This aspect of the study was restricted to Methyl Bay and Granville Lake. Details of the method are presented in Ramsey (1989a).

The methanogens and sulphate reducers were found to jointly account for the majority of seasonal mean methylation (66%) and demethylation (50%) in Granville Lake sediments (Table 1). The two groups were equal contributors to both processes. Conditions in the flooded zone of Methyl Bay appear to be particularly attractive to these groups, where they jointly accounted for all of methylation and demethylation. The share of total methylation and demethylation attributable to the methanogens in flooded sediments was the same as in Granville Lake. The shares of total methylation and demethylation attributable to the sulphate reducers in Methyl Bay sediments were approximately double those in Granville Lake. The rates of methylation and demethylation by methanogens in Methyl Bay were a factor of 5 higher than in Granville Lake. Methylation and demethylation rates by sulphate reducers were a factor of 10 higher

Table 1. Comparison of seasonal mean specific rates of microbial methylation and demethylation attributable to methanogenic and sulphate reducing bacteria; July 4 through August 24, 1988. All values are time-weighted means from Ramsey (1989a).

	% added Hg g^{-1} h^{-1}					
	Methylation			Demethylation		
Site	Total	Methanogen	Sulfate Reducers	Total	Methanogen	Sulfate Reducers
Methyl Bay						
1	0.0034	0.0015	0.0029	0.2620	0.0939	0.1404
2	0.0054	0.0026	0.0033	0.3313	0.1862	0.2034
3	0.0042	0.0003	0.0023	0.1004	0.0127	0.0409
Mean	0.0043	0.0015	0.0028	0.2279	0.0976	0.1282
Granville Lake						
5	0.0010	0.0000	0.0006	0.0861	0.0137	0.0175
6	0.0009	0.0008	0.0001	0.0438	0.0140	0.0034
7	0.0007	0.0001	0.0002	0.0523	0.0221	0.0190
Mean	0.0009	0.0003	0.0003	0.0581	0.0166	0.0133

in the flooded sediments.

Both groups of bacteria are **obligate anaerobes** (Mitchell 1974). The increased absolute rates of methylation and demethylation by these groups, and the increased importance of sulfate reducers in these processes are probably a result of the environmental conditions which develop in the flooded soils and vegetation after flooding. The high rates of **aerobic** decomposition in these organic rich sediments could be expected to create anoxic micro-environments and, in extreme cases, complete anoxia in the sediment pore-waters and near-bottom waters. Methanogenesis requires the complete absence of oxygen while sulphate reduction is less strict in this requirement (Mitchell 1974).

The importance of these groups in methylation explains the finding of higher rates in anoxic sediments along the CRD (Ramsey and Ramlal 1987a; Jackson 1987). Recent research suggests methylation rates are highest in the zone of transition between oxic and anoxic conditions (Winfrey and Rislove 1990). This is consistent with the identity of the methylating organisms, and with the chemistry of inorganic mercury.

An inverse relationship between oxygen availability and methylation rate has been frequently reported (Beijer and Jernelov 1979), but it was difficult to understand how this could occur based on the chemistry of inorganic mercury. The sulphides (S^{2-}) produced during sulphate reduction would be expected to bind with mercuric ions (Hg^{2+}) forming cinnabar (mercuric sulphide, HgS). Under anoxic conditions, cinnabar is not available for methylation

(Beijer and Jernelov 1979). This paradox is avoided in the transitional zone. Sulfate reduction can occur, but inorganic mercury is not rendered completely unavailable.

2.0 Mercury in Water

2.1 Total Mercury

The discovery of elevated mercury concentrations in fish from Southern Indian Lake and the Churchill River diversion suggested the that elevated mercury concentrations might be found in the water as well. The fish were accumulating their increased mercury burdens from the water, both directly over the gills and indirectly in their food. Thus, it seemed logical to look for mercury in the water both as an indication of the cause of the problem in fish as well as to determine if the quality of water for human consumption and the maintenance of aquatic life had been affected.

Several approaches and a variety of analytical methods have been applied to the problem of measuring mercury concentrations in water since 1981. Estimates of concentrations have varied widely among the studies, but this variation was largely a result of inadequate sampling methods which introduced contamination. Despite this variability, all studies have been consistent in concluding that concentrations in the flooded lakes are not elevated in comparison to sites which have not been affected by Churchill River diversion.

Bodaly et al. (1984) reported total mercury concentrations from 17 locations on Southern Indian, Issett, Pemichigamau, East, Central, and West Mynarski, Notigi, and Footprint lakes in samples collected in September, 1978, and July, 1981. All values were below the analytical detection limit of 5 ng L^{-1}. These measurements indicated total mercury levels in water were very low relative to guidelines for drinking water, where the upper limit for consumption is 1,000 ng L^{-1} (Health and Welfare Canada 1989), and for protection of aquatic life. The Canadian water quality guideline for protection of aquatic life is 50 ng L^{-1} as total mercury (CCREM 1987). The Manitoba guideline for protection of aquatic life is 6 ng L^{-1} as total mercury (Williamson 1988).

Although the levels were low, it could not be determined if mercury concentrations in water had increased as a result of impoundment and diversion because all values were below the detection limit. It was possible that mercury concentrations had increased by as much as a factor of 5 over the pre-development level, yet still remained below the detection limit of available methods. Isotope studies in limnocorrals indicated actual total mercury concentrations might be as low as 1 ng L^{-1} (Hecky et al. 1987a). Information on the magnitude of an increase in mercury levels in water, if any had occurred, was still needed and investigators turned to measurements of total mercury in net-plankton as an indirect means of addressing the question. Net-plankton were selected as a surrogate for measurements in water based on research which indicated plankton accumulated total mercury in proportion to the concentration in water (reviewed by Ramsey and Ramlal 1987a). The plankton therefore represented a natural pre-concentration step in the analytical procedure.

Most measurements of mercury concentrations in plankton have indicated that total mercury concentrations in water of SIL, the CRD, and Sipiwesk and Stephens lakes on the Nelson River are not elevated in comparison to levels in the Churchill River upstream of SIL (Ramsey and Ramlal 1987 a and b; Ramsey 1987a, 1988a). Concentrations in plankton from the reservoir sites were consistently lower than in reference lakes like Granville and East Mynarski (Figure 3). There are no pre-development net-plankton mercury data for the flooded lakes, so measurements in non-flooded reference lakes provided the next best indication of baseline levels.

The conclusion of the net-plankton studies was confirmed by direct measurement in 1987, when changes in sample collection procedures enabled the reduction of detection limits. Total mercury concentrations in water were positively correlated with concentrations in net-plankton (Figure 4), and the mercury concentrations in water from SIL and Notigi Reservoir were lower than in Granville Lake, upstream of the flooding (Table 2).

The range of total mercury concentrations in waters from the NFA area is summarized in Table 2. There is considerable variance in reported levels with a general trend of decreasing concentrations over time. This decline was a result of improvements of sampling and analytical methods rather than representing a temporal trend. The potential for contamination of samples, both in the field and in the lab, is considerable (Robertson et al. 1987; Gill

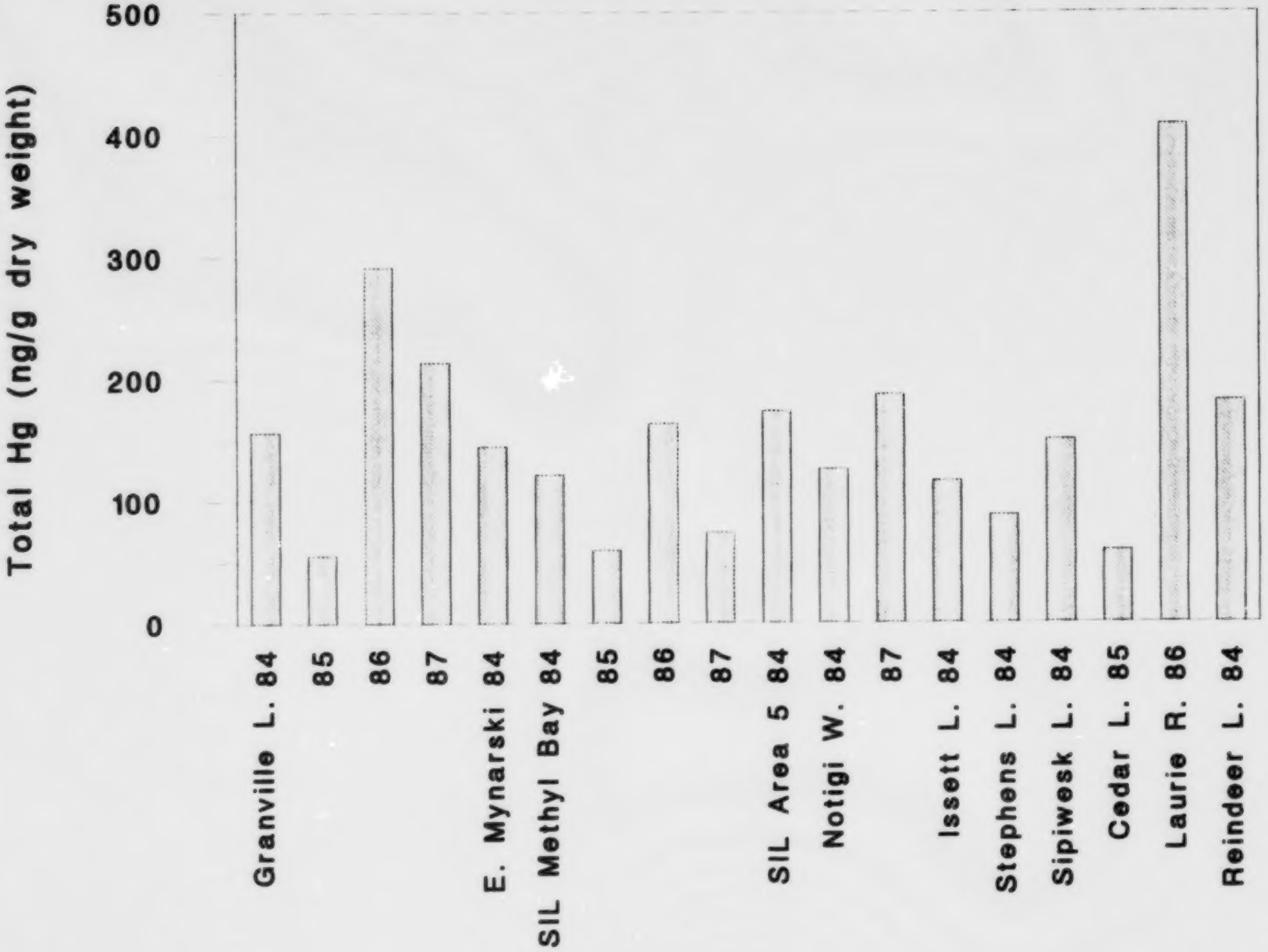


Figure 3. Seasonal mean total mercury concentrations in net-plankton from lakes and reservoirs in northern Manitoba, 1984-1987. Data from Ramsey and Ramlal (1987 a and b), Ramsey (1987a), and Ramsey (1988a).

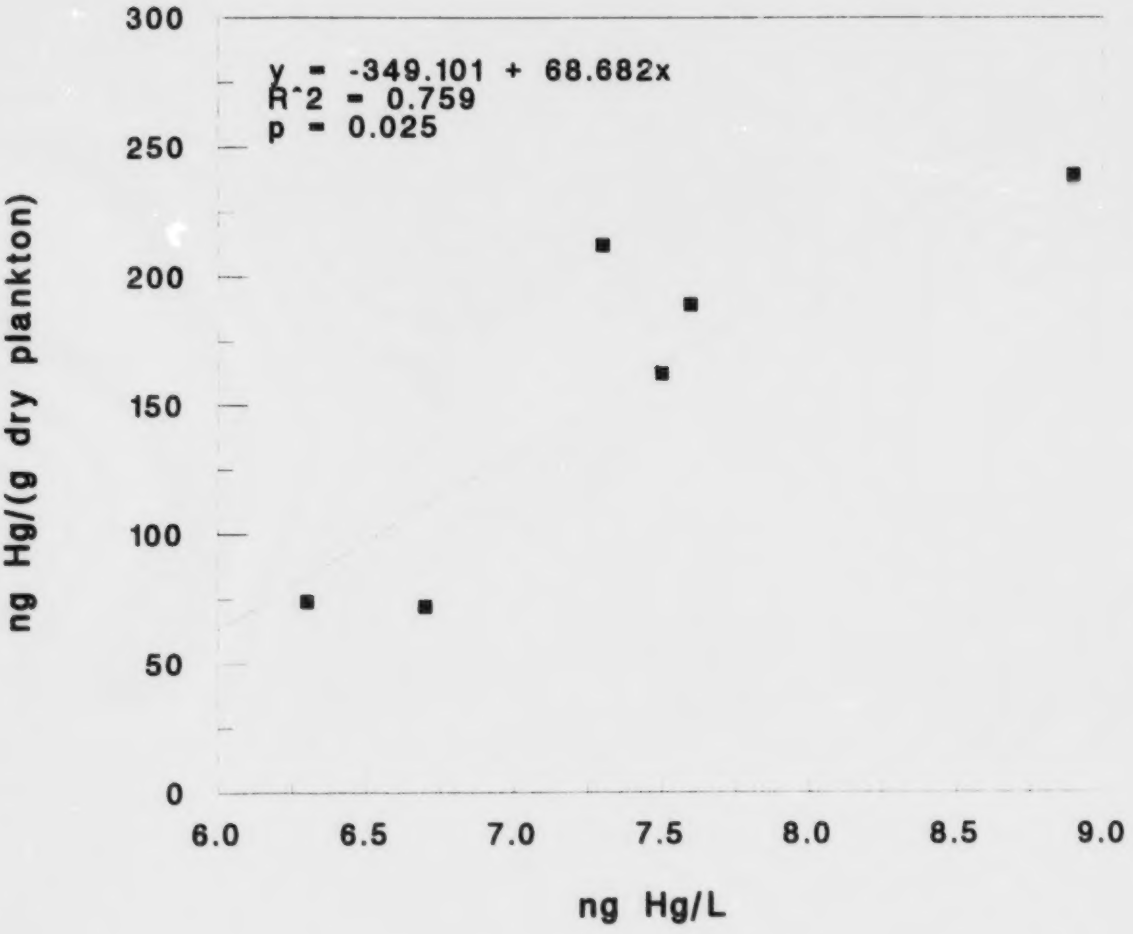


Figure 4. Relationship between seasonal mean total mercury concentrations in water and in net-plankton. From Ramsey (1988a).

Table 2. Total and methyl mercury concentrations in water samples from lakes and reservoirs in northern Manitoba. Data for 1981 from Jackson (1988; Table 10). Data for 1987 and 1989 are from Ramsey (1988a and 1990) and are time-weighted means (with range) for the July-August period.

Year	Lake		Total Hg (ng L^{-1})	Methyl Hg (ng L^{-1})
1981				
	Granville		31.9	0.69 (0.27-1.11)
	Southern Indian			
	Long Bay		23.9	0.29 (0.17-0.40)
	South Bay		26.9	0.30 (0.19-0.41)
	Notigi			
	West		28.3	0.28 (0.38-0.17)
	East	(epilimnion)	21.5	0.60 (0.29-0.91)
	East	(hypolimnion)	36.7	0.59 (0.55-0.63)
	Rat		25.5	0.43 (0.28-0.58)
	Central Mynarski		26.3	0.50 (0.29-0.70)
1987				
	Granville	(offshore)	8.3 (4.5-15.1)	ND
	Southern Indian			
	Methyl Bay	(offshore)	7.2 (5.2-9.0)	ND
	Notigi West	(offshore)	7.4 (4.5-10.3)	ND
1989				
	Granville	(nearshore)	1.44 (0.70-1.82)	0.021 (0.013-0.033)
		(offshore)	1.50 (0.66-2.37)	0.023 (0.020-0.038)
	Southern Indian			
	Methyl Bay	(nearshore)	1.18 (0.71-2.37)	0.034 (0.028-0.045)
		(offshore)	1.60 (0.73-3.15)	0.017 (0.014-0.023)
	Notigi West		1.07	0.046
	Burntwood River at Thompson		2.10	0.027 (0.023-0.031)
	Stephens	(nearshore)	1.77 (1.07-2.70)	0.037 (0.027-0.044)
		(offshore)	1.28 (0.81-1.73)	0.054 (0.041-0.060)

ND No data.

and Fitzgerald 1985). As these sources of contamination are controlled, the measured concentrations decline.

The most important source of contamination appears to have been the sampler used in collection. The highest concentrations (1981 and 1987; Table 2) were measured in samples collected with various types of plastic samplers (i.e., van Dorn bottle or plastic jug, Jackson 1988; van Dorn bottle, Ramsey 1988a), while the low levels measured in 1989 were in samples in which the bottles were filled directly from the lake (Ramsey 1990). Extensive measures also were taken in 1989 to clean the sample bottles before use, to protect the samples from contamination while in the field, and to minimize the mercury content of analytical reagents (Ramsey 1990).

2.2 Methyl Mercury

Although total mercury levels in waters of the CRD have not been elevated as a result of flooding, methyl mercury levels have increased (Ramsey 1990). Until recently, it was necessary to assume that methyl mercury levels were elevated due to the lack of an appropriate analytical method. This assumption was confirmed in 1989 using a newly developed analytical method (Bloom 1989). The methyl mercury level in the flooded zone of Methyl Bay was 55% higher than in Granville Lake while the level outside the flooded zone was just 77% of the Granville Lake value (Table 2). Elevated methyl mercury concentrations also were measured in Notigi Reservoir, near the outflow, and in Stephens Lake on the Nelson River. Levels at these sites were about 34% higher than in the flooded zone of Methyl Bay, probably due to the greater extent of flooding in these reservoirs. Methyl mercury was elevated in the offshore zone of Stephens Lake as well, unlike in Methyl Bay. This was consistent with the occurrence of elevated net microbial methyl mercury production in the offshore zone of Stephens Lake but not in SIL (Ramsey 1990).

Methyl mercury accounts for a very small portion of total mercury in water, in the range of 1-5% (Table 2), so there is considerable room for an increase in methyl mercury before an increase in total mercury can be measured. There are no guidelines for methyl mercury levels in water, but it seems very unlikely that even the highest levels found in the CRD and Nelson River represent a health threat. This is best illustrated by comparison to the amount of mercury consumed from fish. A person eating 500 g (about 1 lb) of whitefish with a total mercury concentration, all as methyl mercury, of 200 ng g^{-1} (i.e., 0.2 $\mu g\ g^{-1}$) consumes 100,000 ng (or 100 μg) of methyl mercury. According to Health and Welfare Canada fish consumption guidelines, unlimited quantities of fish with this mercury level can be eaten. The highest single methyl mercury level measured in water in 1989 was 0.06 ng L^{-1} in Stephens Lake (Table 2). In order to consume the same quantity of methyl mercury directly from the water as in the single meal of whitefish, a person would have to drink approximately 1.7 million litres of water, providing a graphic illustration of the ability of methyl mercury to accumulate in fish tissues.

2.3 Relationship Between Total and Methyl Mercury

A significant positive correlation between the concentrations of methyl and total mercury in water from Southern Indian and Granville lakes and Notigi Reservoir was reported in one of the early studies on the CRD (Jackson 1988). This suggested methyl mercury production might be limited by mercury availability. However, no correlation between the concentrations of methyl and total mercury was found in the FEMP studies (Table 2; Ramsey 1990).

The contradictory findings of these studies may be a result of the different sampling and analytical methods employed. Jackson (1988) used the solvent extraction procedure of Uthe et al. (1972) for the analysis of methyl mercury. This method was originally developed for the analysis of methyl mercury in fish tissues and subsequently adapted for analyses in water. A detection limit of 0.1-0.2 ng L^{-1} was stated but no basis for this claim was provided. The analyses also were performed on centrifuge supernatant rather than whole water, with the centrifugation step representing a potentially serious source of contamination. In the FEMP study, methyl mercury analyses were performed on samples of whole water collected directly from the lake in teflon sample bottles. The sample bottles were cleaned in hot acid before each trip, and were double bagged for transportation and storage. Plastic gloves were worn whenever the bottles were handled. These measures have been found necessary to prevent sample contamination (N. Bloom, Brooks Rand Ltd., Seattle, WA, pers. comm.).

The presence of contamination in Jackson's samples is indicated by concentrations of methyl mercury in water from Granville Lake (0.25 and 1.03 ng L^{-1} as Hg; Table 2) which are 10-100 times higher than found in natural oligotrophic and mesotrophic lakes elsewhere (<0.004-0.039 ng L^{-1}; Bloom 1989). Granville Lake is a pristine lake on the Churchill River upstream from SIL and unaffected by hydroelectric or other industrial developments.

There also was evidence of contamination in Jackson's total mercury samples. The reported concentration in Granville Lake water (32 ng L^{-1} as Hg; Table 2) is substantially higher than measured in other natural waters using clean sampling and analytical techniques (2.8-4.1 ng L^{-1} as Hg; Robertson et al. 1987), and is comparable to levels reported for sites of industrial pollution. Rudd and Turner (1983) reported total mercury concentrations in the range of 18-35 ng L^{-1}, as Hg, in the highly mercury-contaminated Clay Lake, Ontario. Jackson's analytical results clearly are questionable, raising considerable doubt regarding the reported correlation between total and methyl mercury levels in CRD waters. Such a relationship is inconsistent with present knowledge of the factors regulating methyl mercury production in these lakes (Section 3.0). Rates of methyl mercury production and levels in water and fish are not related to total mercury levels, either in source materials or in the water column.

2.4 Downstream Transport of Methyl Mercury

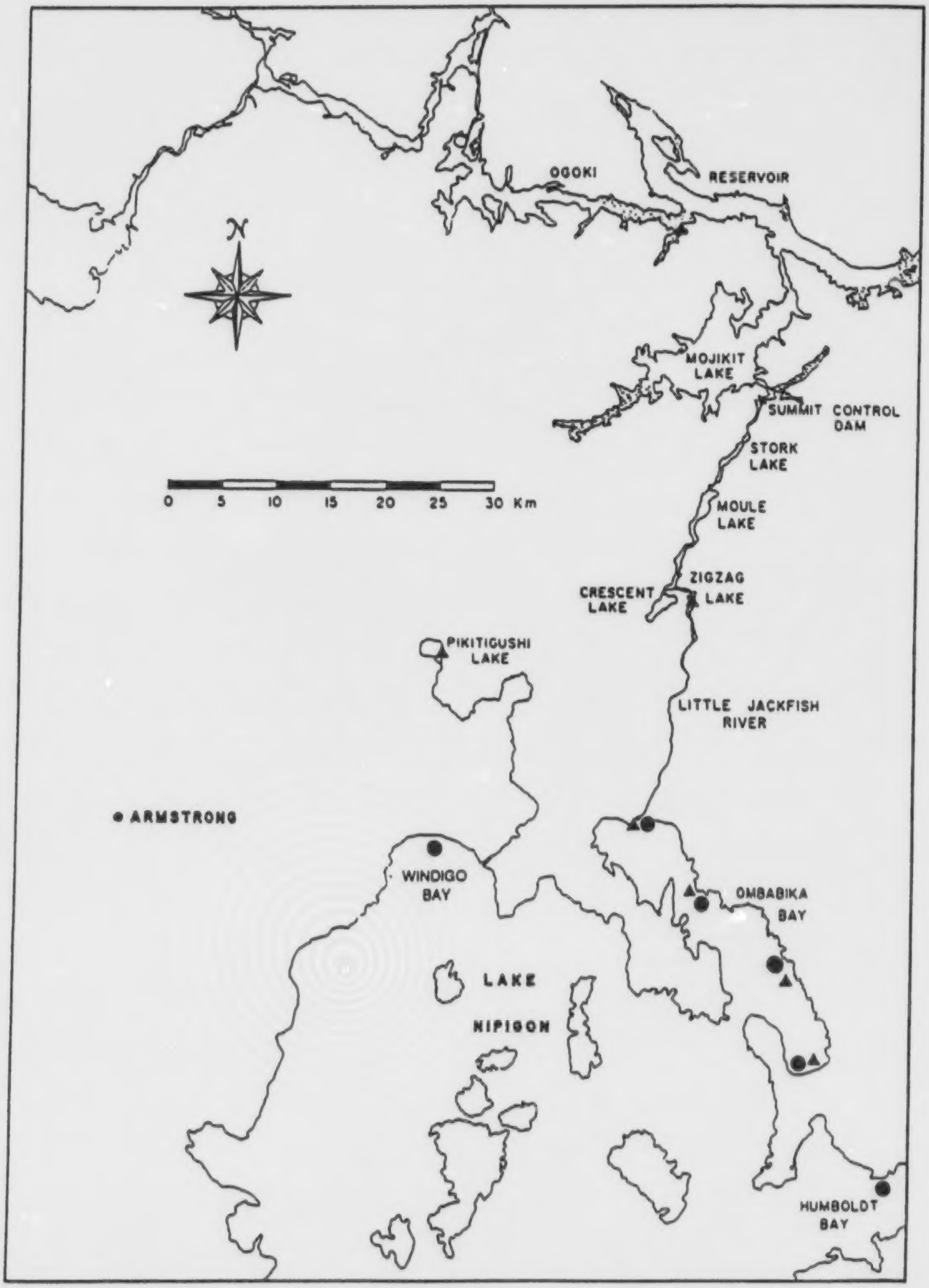
There are a number of indications that elevated methyl mercury production in the extensively flooded areas of the CRD, such as in Notigi Reservoir, may be affecting fish mercury levels in downstream watercourses as well. Derksen and Green (1987) identified 3 lakes on the CRD (Wapisu, Mynarski, and Wuskwatim) where the increases in fish mercury levels following diversion were greater than expected on the basis of the extent of flooding, and suggested downstream transport of methyl mercury from Notigi Reservoir may be responsible. Similarly, Johnston et al. (1991) found the area of flooded land upstream of a lake or reservoir on the CRD was an important factor in determining fish mercury levels.

The higher than expected fish mercury levels downstream of Notigi Reservoir probably are the combined result of movements of mercury contaminated fish downstream through the control structure at Notigi and the discharge of methyl mercury contaminated water. The relative importance of these pathways remains to be determined.

The 1989 survey of methyl mercury levels in CRD waters included measurements in the Burntwood River at Thompson to look for evidence of downstream transport. The range of concentrations at this station was the same as in Granville Lake (Table 2), situated above the reservoirs on the CRD. Although there is considerable evidence to indicate methyl mercury from Notigi Reservoir is transported as far as Wuskwatim Lake, these measurements suggest the methyl mercury does not travel as far as Thompson, 130 km downstream of the Notigi Dam. The actual limit of downstream transport and the mechanism by which the methyl mercury is lost from the water column remain to be examined.

Evidence of downstream effects also has been found on the Ogoki River diversion in northwestern Ontario. Mercury concentrations in spottail shiners from Ombabika Bay on Lake Nipigon, which receives flows from the Ogoki River diversion via the Little Jackfish River (Figure 5), are significantly higher than in fish from other parts of the lake (Figure 6). The highest level is found at the northern end of the bay, at the mouth of the Little Jackfish River, and levels decrease southward with distance from the mouth of the river. The methylation balance in Ombabika Bay is not elevated, indicating the high mercury levels in fish are a product of the downstream transport of methyl mercury from the upstream Ogoki Reservoir (Ramsey 1987b).

Downstream effects have been documented on the La Grande River below the LG-2 Generating Station, where mercury concentrations in resident salmonids have become seriously elevated as a result of the hydro development. The mean concentrations in resident whitefish are 1.02-1.06 $\mu g\ g^{-1}$, compared with 0.38 $\mu g\ g^{-1}$ in anadromous individuals captured in the river and 0.06-0.2 $\mu g\ g^{-1}$ in other natural rivers draining into James Bay (Roy 1990). Levels in whitefish resident in the river below the LG-2 dam also are much higher than in fish from the LG-2 reservoir, where mercury concentrations were 0.40-0.54 $\mu g\ g^{-1}$ between 1984 and 1988. The


OGOKI
RESERVOIR
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MOJIKIT LAKE
SUMMIT CONTROL DAM
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MOULE LAKE
ZIGZAG LAKE
CRESCENT LAKE
PIKITIGUSHI LAKE
LITTLE JACKFISH RIVER
ARMSTRONG
WINDIGO BAY
OMBABIKA BAY
LAKE NIPIGON
HUMBOLDT BAY

Figure 5. Location of fish mercury (●) and methylation balance (▲) sampling sites on Lake Nipigon and the Little Jackfish River system, 1986.

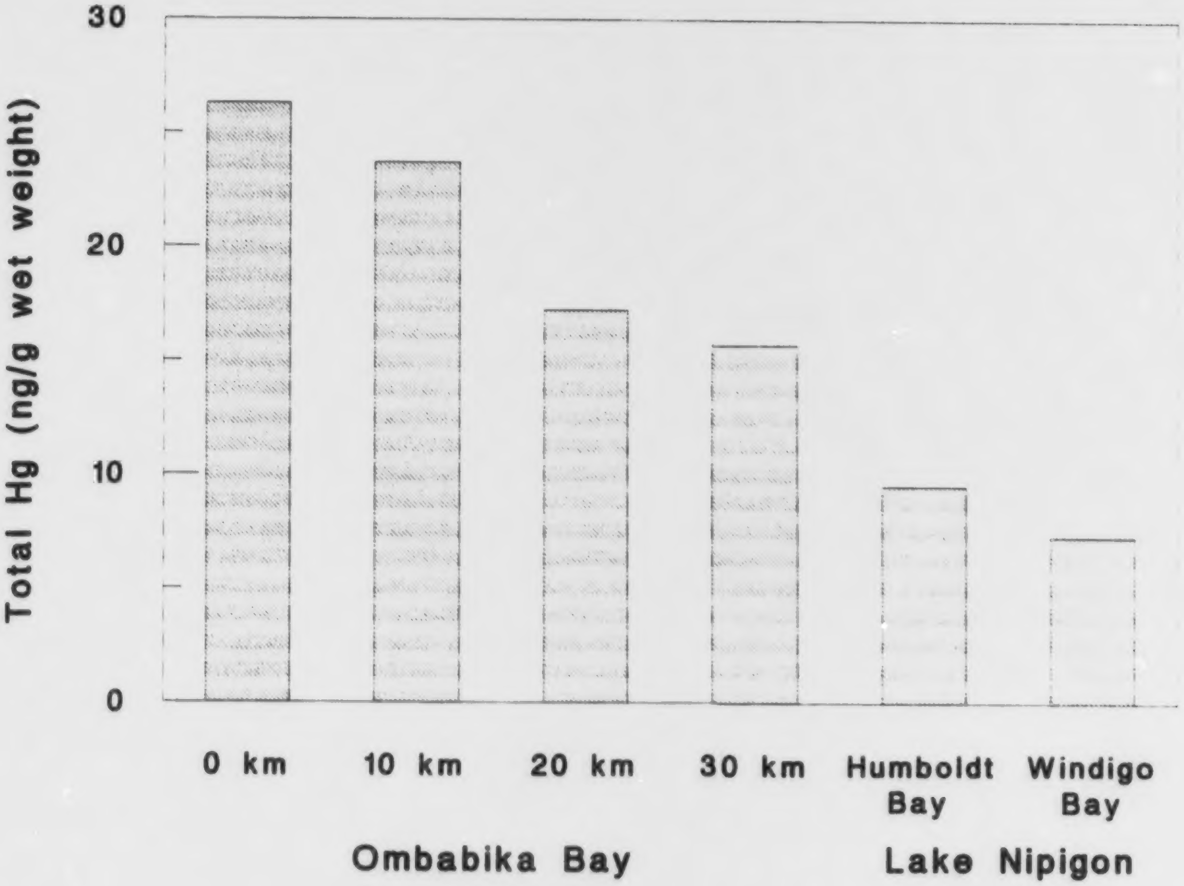

Figure 6. Total mercury concentrations in spottail shiners from sites on Lake Nipigon, August 1986. See Figure 5 for site locations. Ombabika Bay 0 km was at the mouth of the Little Jackfish River. Other stations in Ombabika Bay were the indicated distance from the mouth of the river. Data from Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario.

higher downstream levels are speculated to be a result of the whitefish adopting a predatory feeding habit, consuming the remains of fish which have passed through the LG-2 turbines. This hypothesis remains to be tested, however.

2.4.1 Split Lake

Located at the confluence of the Nelson and Burntwood rivers, and with reservoirs located upstream on both of these major tributaries, Split Lake (Figure 1) would appear to be a likely site to find elevated mercury levels in fish as result of downstream transport. Mercury concentrations in Split Lake fish also are high enough to suggest a possible upstream influence. On average, the standardized mercury level in walleye captured in survey samples was 0.53 $\mu g\ g^{-1}$ between 1983 and 1989 (Ramsey 1991), and 29% of the walleye catch had a mercury concentration in excess of 0.54 $\mu g\ g^{-1}$ (Green 1990). The mean standardized mercury level in pike was 0.38 $\mu g\ g^{-1}$, and 20% of the survey catch exceeded 0.54 $\mu g\ g^{-1}$ during the same period (Green 1990). Because these levels exceed 0.2 $\mu g\ g^{-1}$, some residents of the Split Lake community with blood mercury levels in excess of 20 ppb have been advised by Health and Welfare Canada to avoid the consumption of pike and walleye as a staple in their diet (B. Stiff, Health and Welfare Canada, Winnipeg, MB, pers. comm.). This restriction has caused concern in the communities and raised questions regarding the effect of past hydro development on fish mercury levels in the lake.

There is no means of directly proving that fish mercury levels in Split Lake are elevated in the absence of pre-development data. The levels in pike and walleye also are not unusual compared to the natural range in northern Manitoba (Derksen and Green 1987). However, the level in walleye in particular appears to be uncharacteristically high for lakes on the Nelson and Burntwood river systems which either have not been flooded, or prior to flooding by hydro developments. The mean mercury levels in survey samples of walleye and northern pike from Playgreen Lake in 1978 and 1981 were 0.20 and 0.21 $\mu g\ g^{-1}$, respectively, and no more than 2% of the catch of either species exceeded 0.54 $\mu g\ g^{-1}$ (Rannie and Punter 1987). The ranges of mean levels in walleye and pike from Cross Lake (west basin) were 0.28-0.33 and 0.31-0.40 $\mu g\ g^{-1}$, respectively, between 1987 and 1989. Less than 6% of the walleye catch exceeded 0.54 $\mu g\ g^{-1}$ (Green 1990). Walleye mercury in Wuskwatim Lake on the Burntwood River averaged 0.36 $\mu g\ g^{-1}$ in commercial samples prior to flooding by the Churchill River diversion (Bodaly et al. 1984).

The seemingly high mercury levels in walleye and pike are not related to Churchill River diversion as they were prevalent before the diversion began operating (Figure 7). The measurements of methyl mercury in water made in 1989 also indicate methyl mercury produced in the Southern Indian Lake and Notigi reservoirs does not travel as far as Thompson and, therefore, could not be affecting fish mercury levels in Split Lake, farther downstream. Mercury concentrations in Split Lake whitefish are negatively correlated

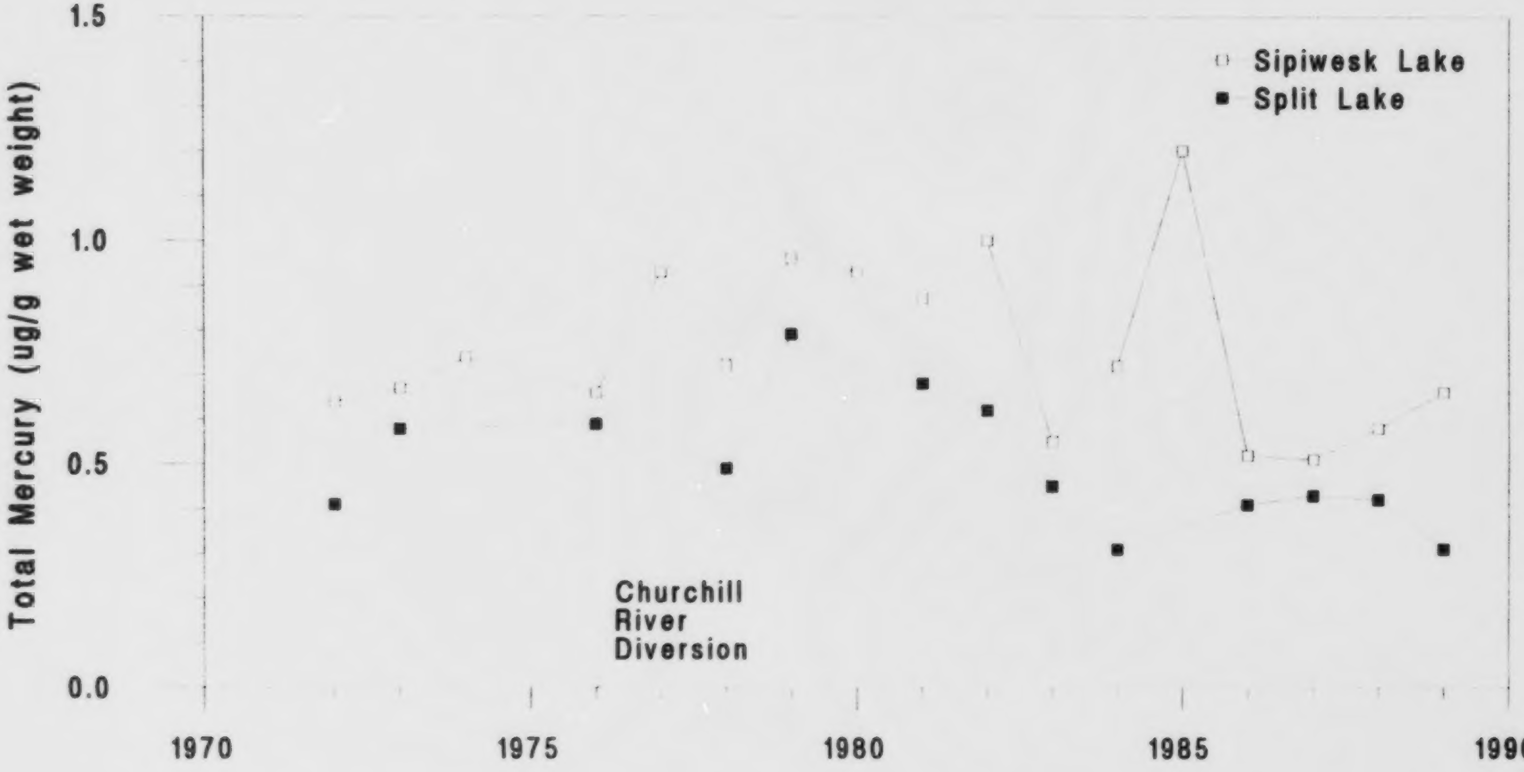

Figure 7. Mean total mercury concentrations in northern pike from Split and Sipiwesk lakes, 1972-1989. Data to 1982 from Derksen and Green (1987). Later data from Green (1990). All data are from lake survey samples.

with Burntwood River discharges (Figure 8), suggesting the waters of the CRD have some form of mitigating influence on methyl mercury availability or uptake in the lake.

It remains to be determined if the methylation balance in Split Lake is naturally high, but this seems to be an unlikely explanation. An external methyl mercury source other than the Nelson River also is improbable. In either case, a negative correlation between mercury levels in whitefish and Nelson River discharge would be expected, because the Nelson River is the major inflow and, therefore, the major means of dilution. However, there is no correlation between mercury in whitefish and Nelson River discharge (Ramsey 1991). This suggests there is a methyl mercury source situated upstream on the Nelson River. Sipiwesk Lake, the Kelsey Dam forebay, and fish killed in the Kelsey Dam turbines are potential sources. In 1987, the microbial mercury methylation balance in the flooded zone of Sipiwesk Lake was elevated compared to the balance in pre-existing lake sediments and a reference lake (Ramsey 1988a). Mercury levels in Split Lake pike also are positively correlated with levels in Sipiwesk Lake pike (Figure 9), further suggesting a direct connection between mercury levels in Split Lake fish and flooding upstream on the Nelson River. The Kelsey forebay has not been sampled but, based on surveys of other reservoirs of equal or greater age, any flooded soils and vegetation probably continue to stimulate mercury methylation there as well. The possibility of methyl mercury becoming more available in Split Lake as a result of fish passing through the turbines of

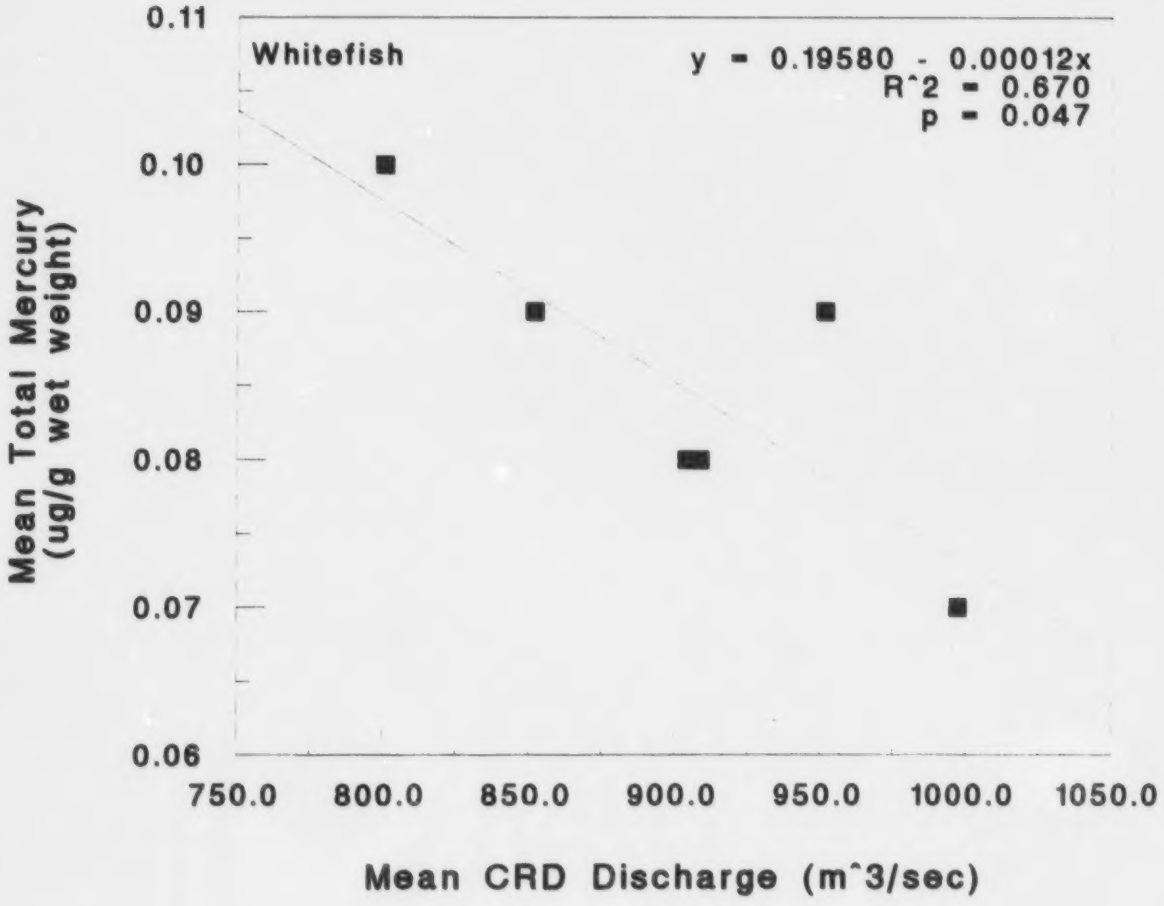

Figure 8. Relationship between length-standardized (350 mm) mercury concentrations in Split Lake whitefish and mean Burntwood River discharge. Discharge data (Water Survey of Canada 1990) averaged over the June to September period in each year. From Ramsey (1991).

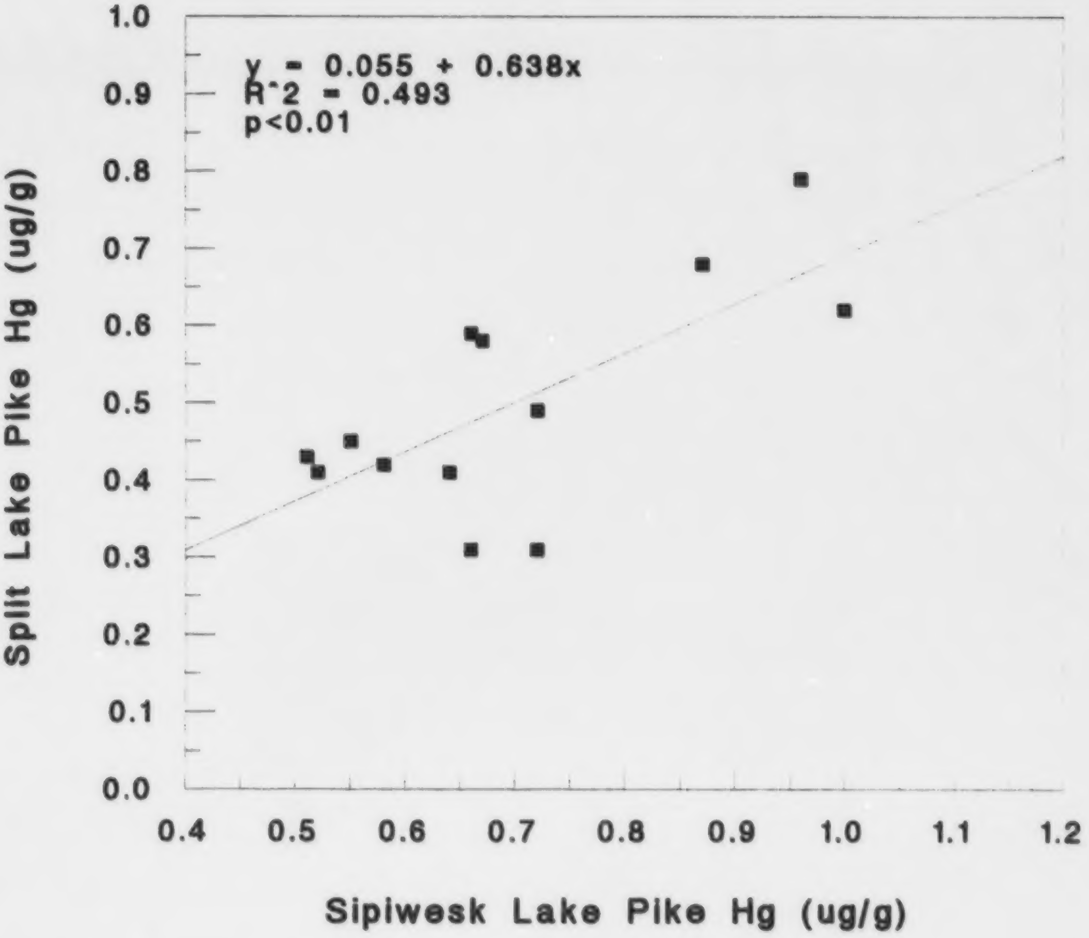

Figure 9. Relationship between mean mercury concentrations in northern pike from Split and Sipiwesk lakes. Data from Figure 7.

the Kelsey Generating Station should be examined as well.

The present significance of hydro developments upstream on the Nelson River as a methyl mercury source to Split Lake could be evaluated using measurements of methyl mercury in water. A series of measurements on the Nelson River between Lake Winnipeg and Split Lake would determine if methyl mercury levels in the river are unusually high and if methyl mercury is being generated at a specific site. The relative importance of other sources (e.g., the Aiken and Burntwood rivers) could be determined in a similar manner. This approach would quantify any continuing contribution of hydroelectric developments on the upper Nelson River to methyl mercury availability in Split Lake.

3.0 Factors Regulating Methylation and Demethylation

3.1 Methylation

3.1.1 Mercury and Organic Matter

The role of the mercury contained in flooded and eroding soils and vegetation in microbial methylation, and its relationship to mercury levels in fish, has been the subject of considerable study on the CRD. This emphasis appears to be a result of the seemingly logical connection between increased mercury loading and the elevation of mercury concentrations in fish. Indeed, this is the approach used for most pollutants. However, it must be kept in mind that the mercury which is accumulating in fish is methyl mercury, while inorganic mercury species predominate in the soils and vegetation. The inorganic mercury in these materials must first be methylated in order for it to move into the fish. The only means by which an increase in inorganic mercury loading could cause an increase in fish mercury levels is if the mercury occurs in a form which is readily available for methylation and if methylation is limited by mercury availability. In this regard, mercury is unlike other pollutants.

There are several indications that microbial methylation, at least in the sediments of lakes and reservoirs in northern Manitoba, is not limited by inorganic mercury availability. Rannie and Punter (1987), using data from 42 lakes in the area of the Churchill River diversion, found there was no relationship between total mercury concentrations in lake sediments and mercury concentrations in northern pike. Bodaly et al. (1987) also found there was no relationship between the mercury concentration in flooded soils from a number of sites on the CRD and mercury concentrations in forage fish (yellow perch, spottail and emerald shiners) from the same areas.

A consistent finding of the methylation balance studies conducted under the FEMP has been a positive correlation between microbial methylation and the concentrations in sediments of both organic carbon and total mercury (Figures 10 and 11). The mercury and organic carbon levels are strongly correlated with each other (i.e., they are collinear) as well (Figure 12). Although the correlation between methylation and sediment mercury could be interpreted as an indication of the importance of mercury in regulating methylation, evidence from the CRD area and elsewhere

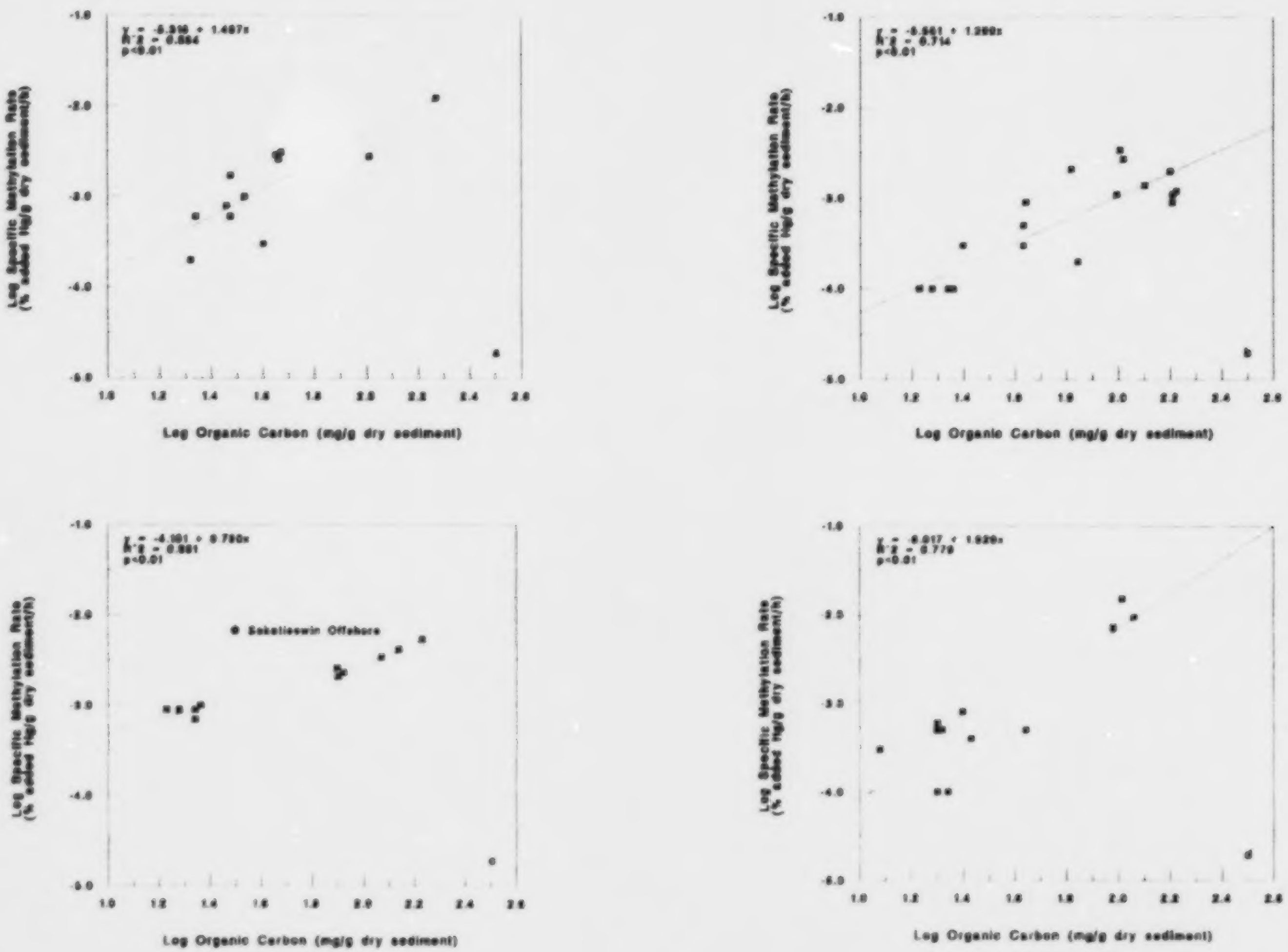

Figure 10. Relationship between seasonal mean organic carbon concentrations and specific methylation rates measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data for Sokatisewin offshore station excluded from regression in (c). Data from Ramsey (1987a, 1988a, 1989a, and 1990).

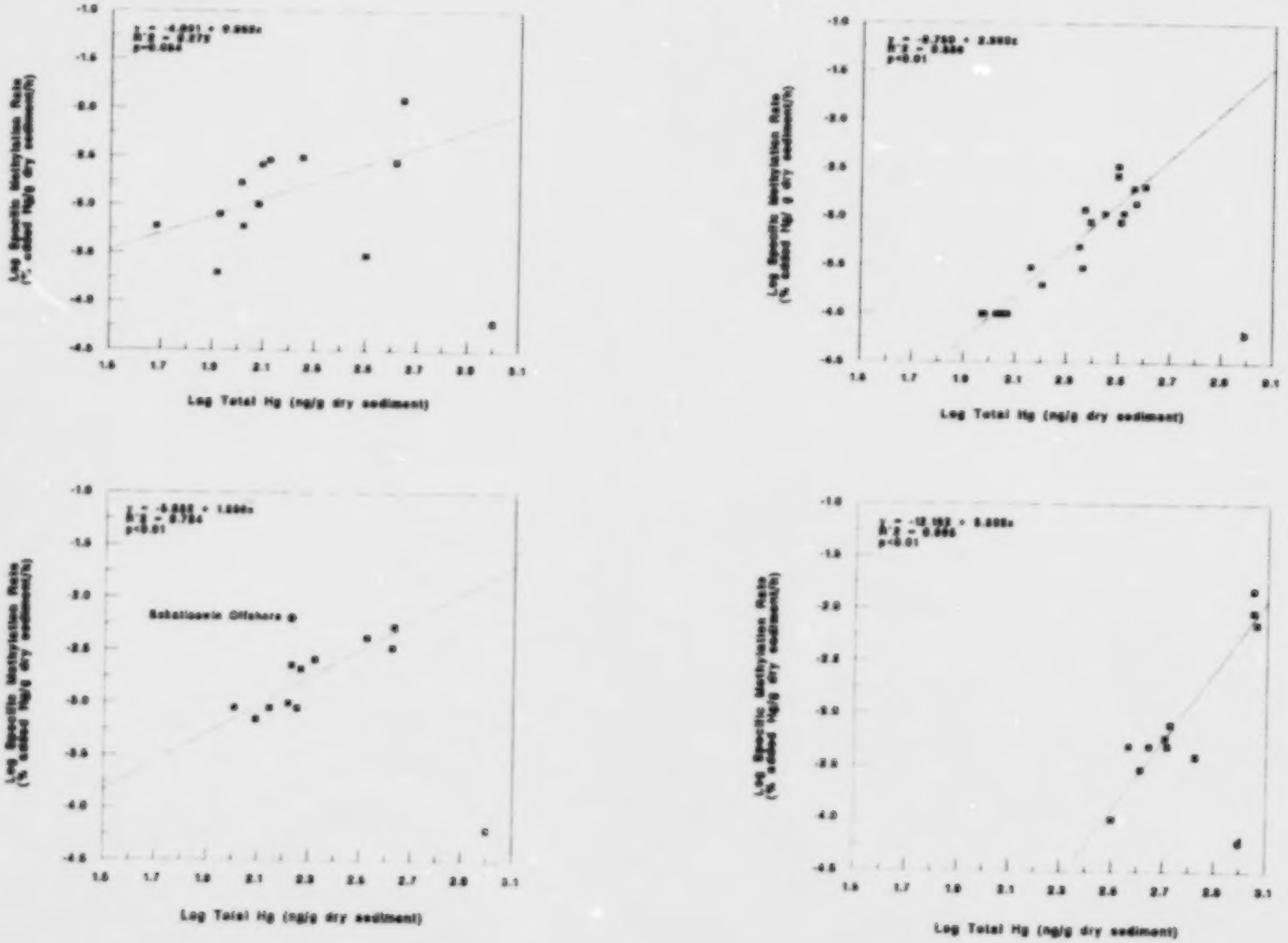

Figure 11. Relationship between seasonal mean total mercury concentrations and specific methylation rates measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data for Sokatisewin offshore station excluded from regression in (c). Data from Ramsey (1987a, 1988a, 1989a, and 1990).

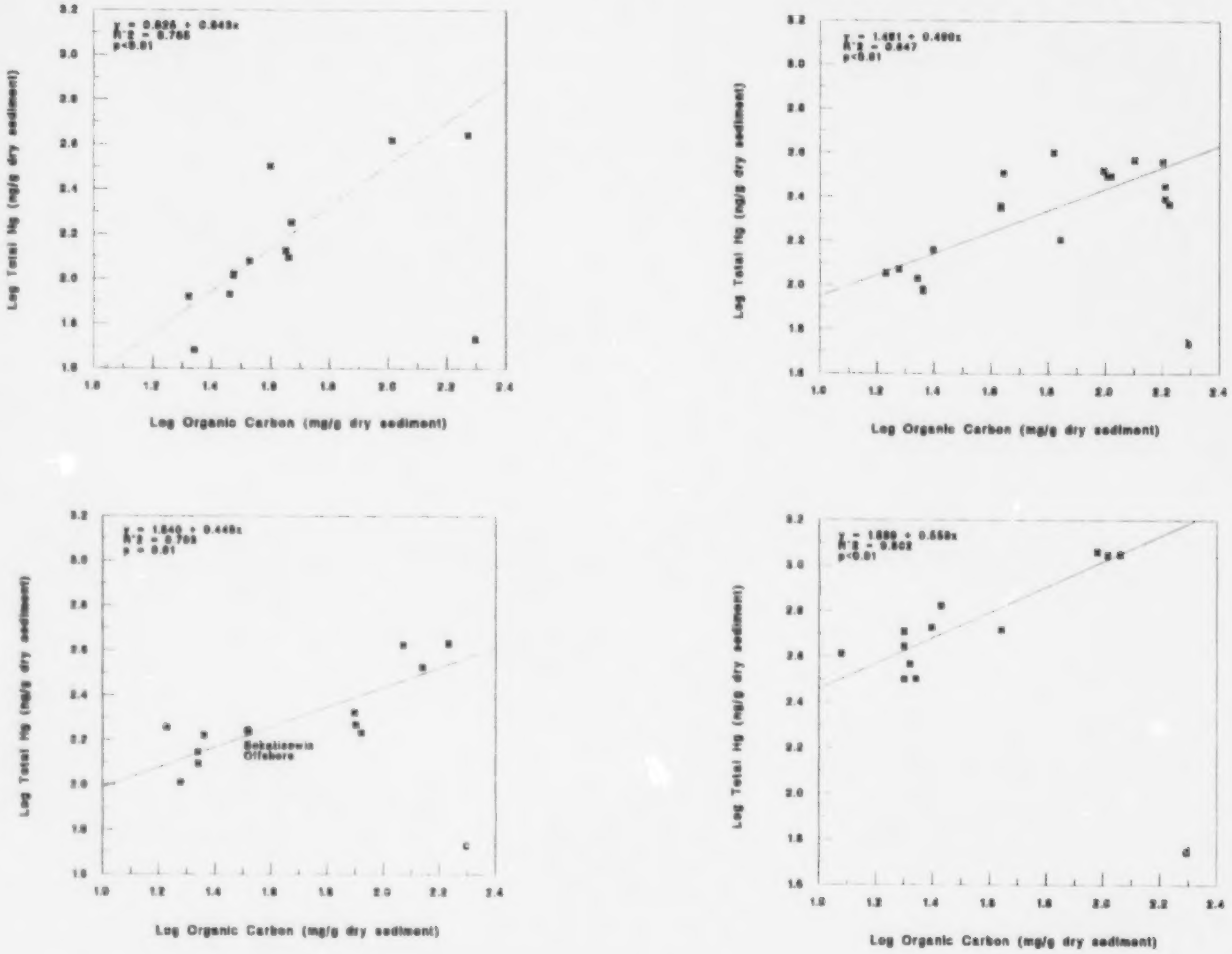

Figure 12. Relationship between seasonal mean organic carbon and total mercury concentrations measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data from Ramsey (1987a, 1988a, 1989a, and 1990).

suggests the relationship is just a product of the collinearity between mercury and organic carbon. When the correlation between organic carbon and total mercury is removed, either through selection of a subset of the data or sampling in a region where mercury and organic carbon are not correlated, the relationship between mercury and methylation disappears, leaving the correlation between organic carbon and mercury intact. This evidence is detailed below.

In 1986, methylation and demethylation rates were measured at three locations: Methyl Bay on SIL, Laurie Reservoir #2 on the Laurie River, and Granville Lake (Figure 1). Four stations were sampled at each site. A marginally significant correlation was found between specific methylation rates and total mercury in sediments when all sites sampled in 1986 were considered together (Figure 11), but this relationship broke down entirely when the data for Methyl Bay were considered separately. Methylation rates at the four stations in Methyl Bay were positively correlated with organic carbon, but not with total mercury in the absence of a correlation between mercury and organic carbon at the four stations (Figure 13). The significant correlation between methylation and total mercury when all sites were considered together also was strongly influenced by a single station, Methyl Bay station 2 (Figure 11). The correlation disappears when this point is removed from the analysis.

In areas where concentrations of total mercury and organic carbon in sediments vary independently, microbial methylation is

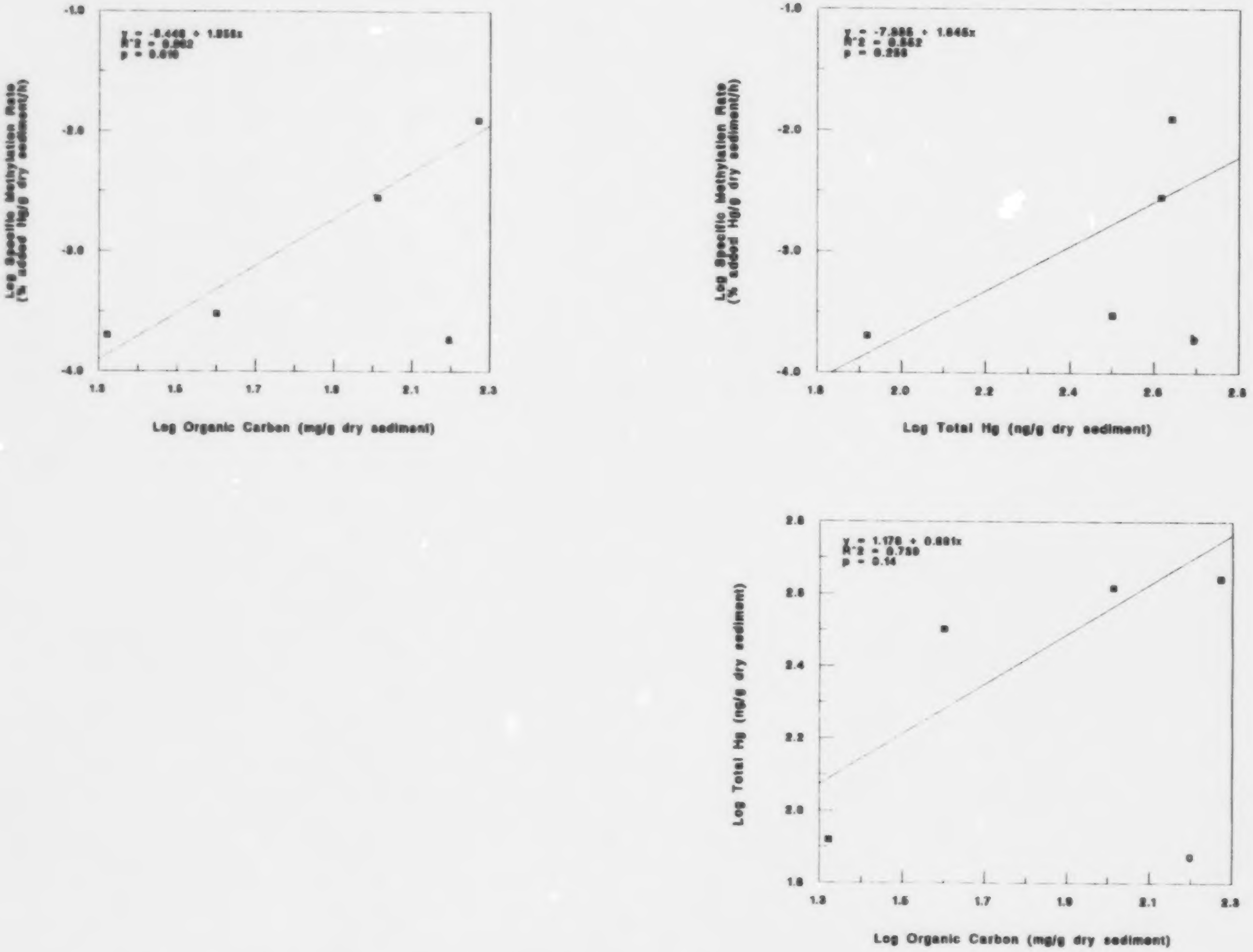

Figure 13. Relationship between (a) seasonal mean total mercury concentrations and specific methylation rates, (b) seasonal mean organic carbon concentrations and specific methylation rates, and (c) seasonal mean organic carbon and total mercury concentrations in sediments from Methyl Bay, Southern Indian Lake, 1986. Data from Ramsey (1987a).

found to be correlated with organic carbon rather than mercury. This is the case in the Little Jackfish River system which flows into Lake Nipigon in northwestern Ontario (Figures 5 and 14), and in streams of the Queen Charlotte Islands (Tables 3 and 4).

Field experiments also have indicated microbial methylation in sediments is not limited by mercury availability. An experiment was conducted on Graham Island, in the Queen Charlotte Islands, British Columbia, in 1988, which examined the influence of soils with a range of total mercury and organic matter contents on microbial methylation and demethylation activity. The experiment involved the submersion of selected soils, contained in plastic basins, in a wetland for approximately 5 weeks before sampling for measurement of specific methylation and demethylation rates using the same radiochemical methods routinely employed in the FEMP studies. Methylation and demethylation rates in the inorganic soils (i.e., soils with less than 20% organic content as measured by loss on ignition; Brady 1974) were similar to the rates in Granville Lake in 1988 (Tables 5 and 6). Methylation and demethylation were independent of mercury content, up to 550 ng g^{-1}, the highest concentration in the experiment. Conversely, organic soil (≥20% organic content) with low mercury content (71 ng g^{-1}, 45% lower than in Granville Lake), supported a specific methylation rate comparable to the mean methylation rate measured at the nearshore stations in Methyl Bay in 1988.

The field studies of microbial methylation balance and mercury levels in fish on the CRD indicated that neither was dependent upon

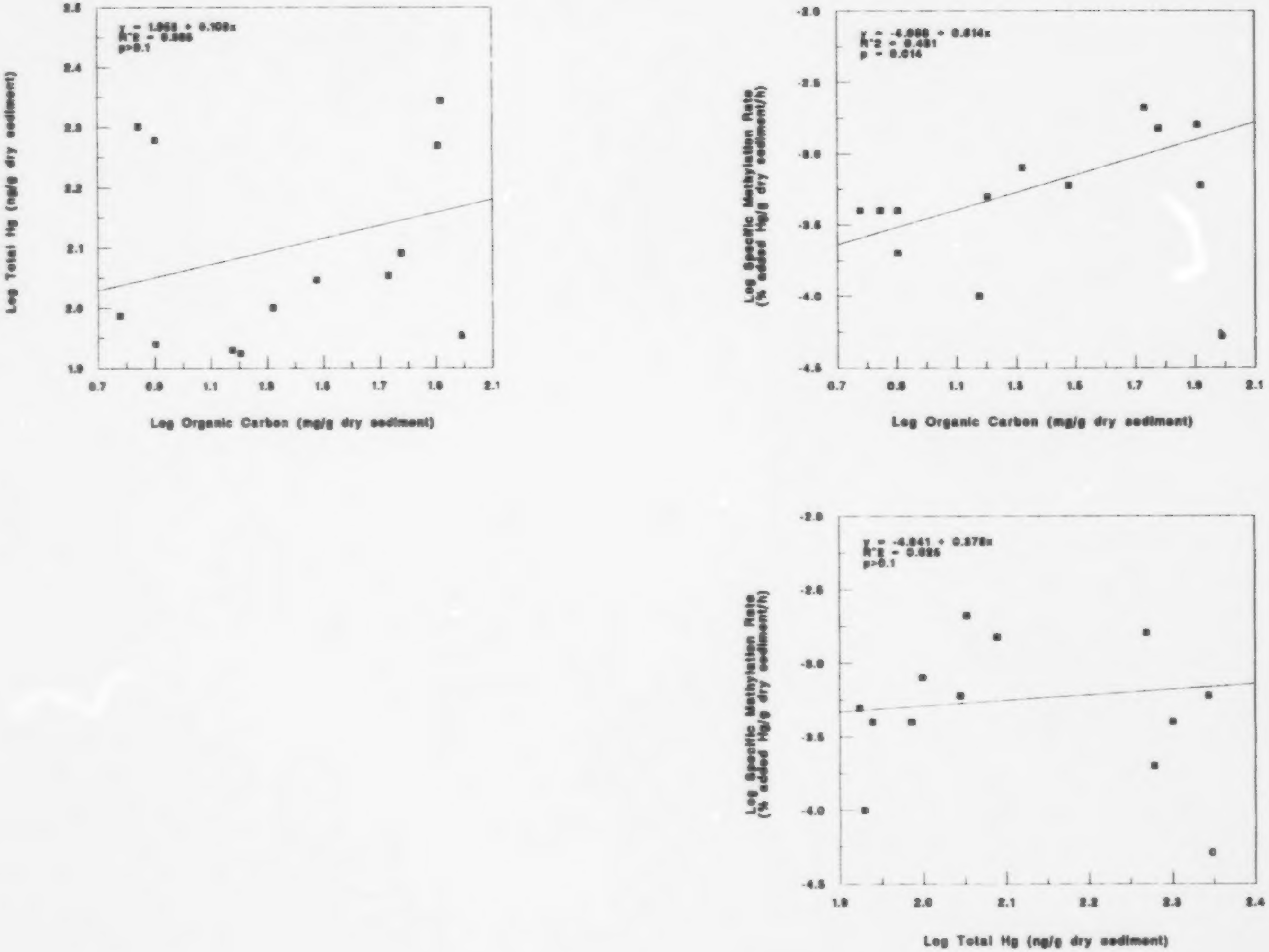

Figure 14. Relationship between (a) seasonal mean organic carbon and total mercury concentrations, (b) seasonal mean organic carbon concentrations and specific methylation rates, and (c) seasonal mean total mercury concentrations and specific methylation rates in sediments from sites on the Little Jackfish River system, northwestern Ontario. Data from Ramsey (1987b).

Table 3. Comparison of specific rates of biological methylation (M), demethylation (D), and methylation balance (M/D) in Graham Island, British Columbia, stream sediments, 1988. Also listed are seasonal mean and peak M, D, and M/D in flooded forest sediments from Southern Indian Lake, Manitoba, and in Granville Lake, Manitoba, an upstream reference lake. Stream data from Ramsey (1988b). Manitoba data from Ramsey (1989a).

			September 20-22	
Site	Sample	M	D	M/D
Middle Barbie Creek	1	<0.001	0.192	<0.005
	2	<0.001	0.362	<0.003
	3	<0.001	0.405	<0.002
	4	<0.001	0.657	<0.002
	Mean	<0.001	0.404	<0.003
Barbie Wetland	1	0.131	0.233	0.562
	2	0.100	0.350	0.286
	3	0.059	0.318	0.186
	4	0.190	0.612	0.310
	Mean	0.120	0.378	0.336
Lower Barbie Creek	1	0.009	0.144	0.063
	2	0.002	0.142	0.014
	3	0.007	0.051	0.137
	4	0.005	0.060	0.083
	Mean	0.006	0.099	0.074
Florence Creek	1	0.001	0.443	0.002
	2	0.001	0.469	0.002
	3	0.003	0.026	0.115
	4	0.002	0.053	0.038
	Mean	0.002	0.248	0.039
Gold Creek	1	0.008	0.026	0.308
	2	0.004	0.177	0.023
	3	0.011	0.083	0.133
	4	0.017	0.098	0.173
	Mean	0.010	0.096	0.159
Southern Indian Lake (July-August 1988)	Peak	0.015	0.540	0.362
	Mean	0.004	0.231	0.017
Granville Lake (July-August 1988)	Peak	0.006	0.394	0.054
	Mean	<0.001	0.058	<0.017

Table 4. Chemical composition of Graham Island stream sediments; September 20-22, 1988. Listed values are the mean and standard deviation (in parenthesis) of analyses on 4 samples. Data from Ramsey (1988b).

Soil Type	Loss on Ignition	Total C	Organic C	Total N	Total P	Total Hg
	(% at 550°C)	(mg g^{-1} dry wt.)				(ng g^{-1} dry wt.)
Middle Barbie	22.5 (4.9)	89 (25)	85 (25)	3.1 (0.5)	0.7 (0.1)	289 (57)
Barbie Wetland	56.4 (0.4)	275 (10)	269 (10)	14.4 (0.6)	1.5 (0.1)	216 (21)
Lower Barbie	22.2 (5.3)	96 (27)	92 (27)	4.7 (1.1)	1.0 (0.1)	247 (25)
Gold Creek	33.6 (3.2)	172 (12)	166 (13)	9.4 (0.4)	1.2 (0.1)	281 (12)
Upper Florence	26.3 (3.6)	110 (5)	104 (4)	5.0 (0.5)	1.2 (0.1)	165 (3.6)

Table 5. Chemical composition of soils tested for use as substrates in the constructed wetlands and water storage reservoir. Ranges list results of two determinations on each soil-type, otherwise values are a single determination. Data from Ramsey (1988b).

Soil Type	Loss on Ignition	Total C	Organic C	Total N	Total P	Total Hg
	(% at 550°C)	(mg g^{-1} dry wt.)				(ng g^{-1} dry wt.)
MSW1	12.0 - 12.6	49	45	1.3	0.4	108 - 125
MSW2-P	17.0 - 17.6	91	86	2.7	0.8	233 - 234
Pit	17.1 - 17.4	113	109	2.7	0.5	466 - 560
Mudstone	25.6 - 28.4	182	180	2.8	0.2	71 - 71
Water Storage	7.9 - 8.2	24	21	0.8	1.0	63 - 69

Table 6. Comparison of specific rates of biological methylation (M), demethylation (D), and methylation balance (M/D) in Graham Island test-soils. Data from Ramsey (1988b).

	MSW1			MSW2-P			Pit			Mudstone			Water Storage		
Sample	M	D	M/D	M	D	M/D	M	D	M/D	M	D	M/D	M	D	M/D
1	<0.001	0.128	<0.008	<0.001	0.009	<0.111	<0.001	0.268	<0.003	0.001	0.098	0.010	<0.001	0.267	<0.004
2	<0.001	0.179	<0.006	<0.001	0.128	<0.009	<0.001	0.025	<0.040	0.002	0.178	0.011	0.001	0.064	0.016
3	<0.001	0.129	<0.008	<0.001	0.008	<0.125	<0.001	0.015	<0.067	0.013	0.046	0.283	<0.001	0.056	<0.018
4	<0.001	0.074	<0.014	<0.001	0.104	<0.010	<0.001	0.027	<0.037	0.009	0.018	0.500	<0.001	0.191	<0.005
Mean	<0.001	0.128	<0.009	<0.001	0.062	<0.064	<0.001	0.112	<0.037	0.006	0.085	0.201	<0.001	0.145	<0.011

the concentration of total mercury in the flooded soils and vegetation, but not why mercury content was unimportant. The mercury in the flooded materials may not be available for methylation, or the supply of available mercury may not be the primary limiting factor of methylation. This question was answered by the limnocorral experiments conducted at SIL between 1981 and 1986 under the CMMA. Hecky et al. (1987a) found that flooded organic matter increased the pool of mercury available for methylation, but the quantity of mercury which accumulated in fish was independent of the size of this pool. This suggested methylation by the natural microbial communities was not limited by inorganic mercury availability. The elevated mercury bioaccumulation by fish in response to the addition of organic soils and vegetation to limnocorrals, and the positive correlation between microbial methylation and sediment organic content in many different environments suggest organic matter availability is the primary limiting factor of methylation in boreal freshwaters.

3.1.2 Microbial Activity and Redox Potential

Organic matter is thought to regulate microbial methylation by controlling microbial activity. The relationships between methylation, demethylation, and microbial activity were examined in the 1988 survey of Methyl Bay, Sokatisewin Lake, and Granville Lake (Ramsey 1989a). Total microbial activity, as indicated by the rate of dissolved inorganic carbon (DIC) production, was measured routinely along with methylation and demethylation.

Contrary to the hypothesis, methylation rates varied independently of microbial activity (Figure 15). Microbial activity and methylation were both elevated in the flooded zone of Methyl Bay, as expected, but elevated methylation rates also occurred in Sokatisewin Lake, where microbial activity was much lower than in Granville Lake.

The explanation for this discrepancy may be found in another characteristic of the Sokatisewin Lake sediments. Oxygen availability in these sediments, as indicated by the oxidation-reduction (redox) potential measured with a platinum electrode, was much lower than in sediments from either Methyl Bay or Granville Lake (Figure 16). Redox potential was lower at the offshore station than nearshore, and the methylation rate was higher offshore. By itself, redox potential was no better predictor of methylation than was microbial activity. However, when the two variables were included in a bivariate regression model, a significant component of the variation in methylation was accounted for ($\log_{10}(M) = 6.975 + 0.998*\log_{10}(MACT) - 2.832*\log_{10}(REDOX)$; $n=12$, $r^2=0.56$, $p=0.025$; where M is the specific methylation rate in % added Hg methylated (g dry sediment)$^{-1}$ h^{-1}, MACT is the rate of DIC production in μmole (g dry sediment)$^{-1}$ h^{-1}, and REDOX is the oxidation-reduction potential in mV).

The importance of redox potential in regulating methylation activity is not surprising given that obligate anaerobic bacteria are responsible for methylation in flooded sediments in Methyl Bay (Section 1.2.4). The activity of these groups should be expected to

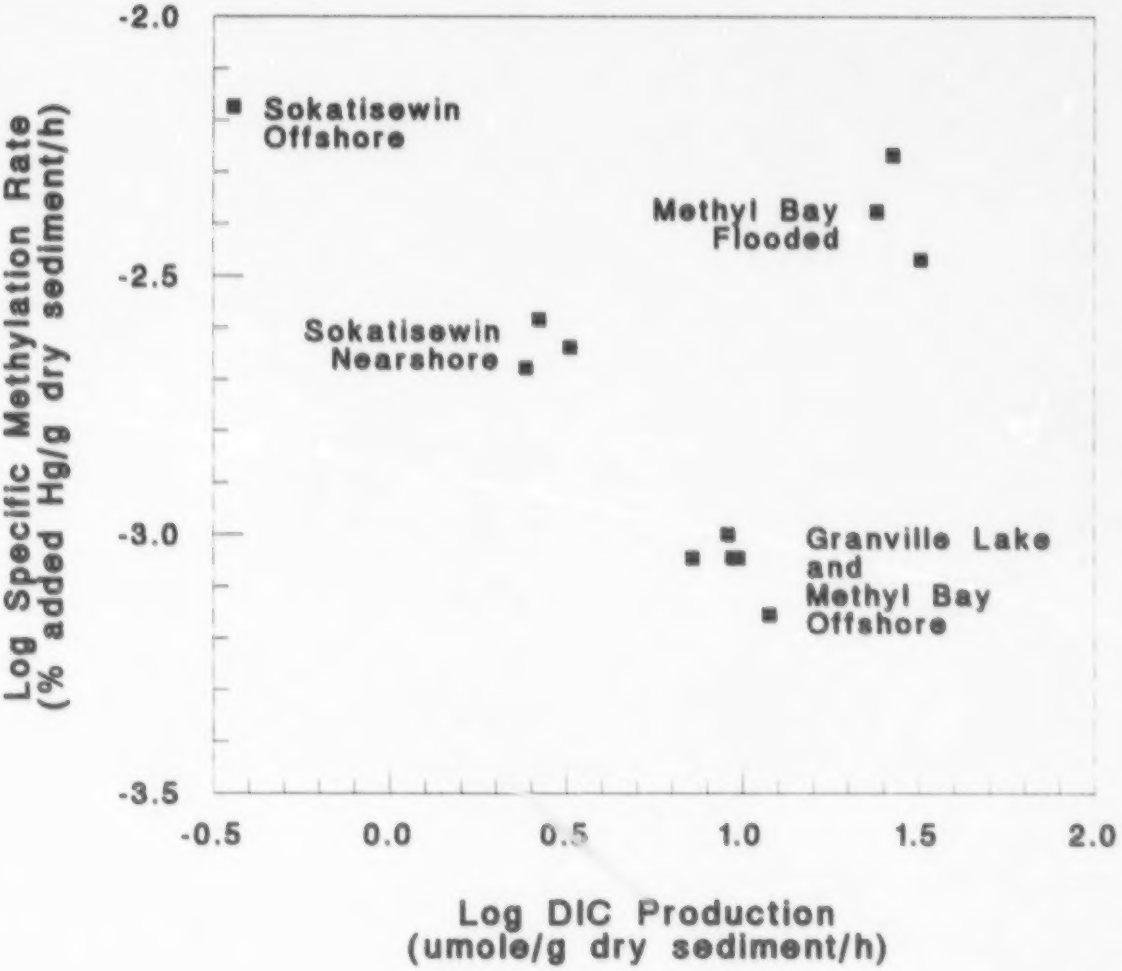

Figure 15. Relationship between the seasonal mean specific methylation rate and microbial activity, as measured by the rate of dissolved inorganic carbon (DIC) production, in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. From Ramsey (1989a).

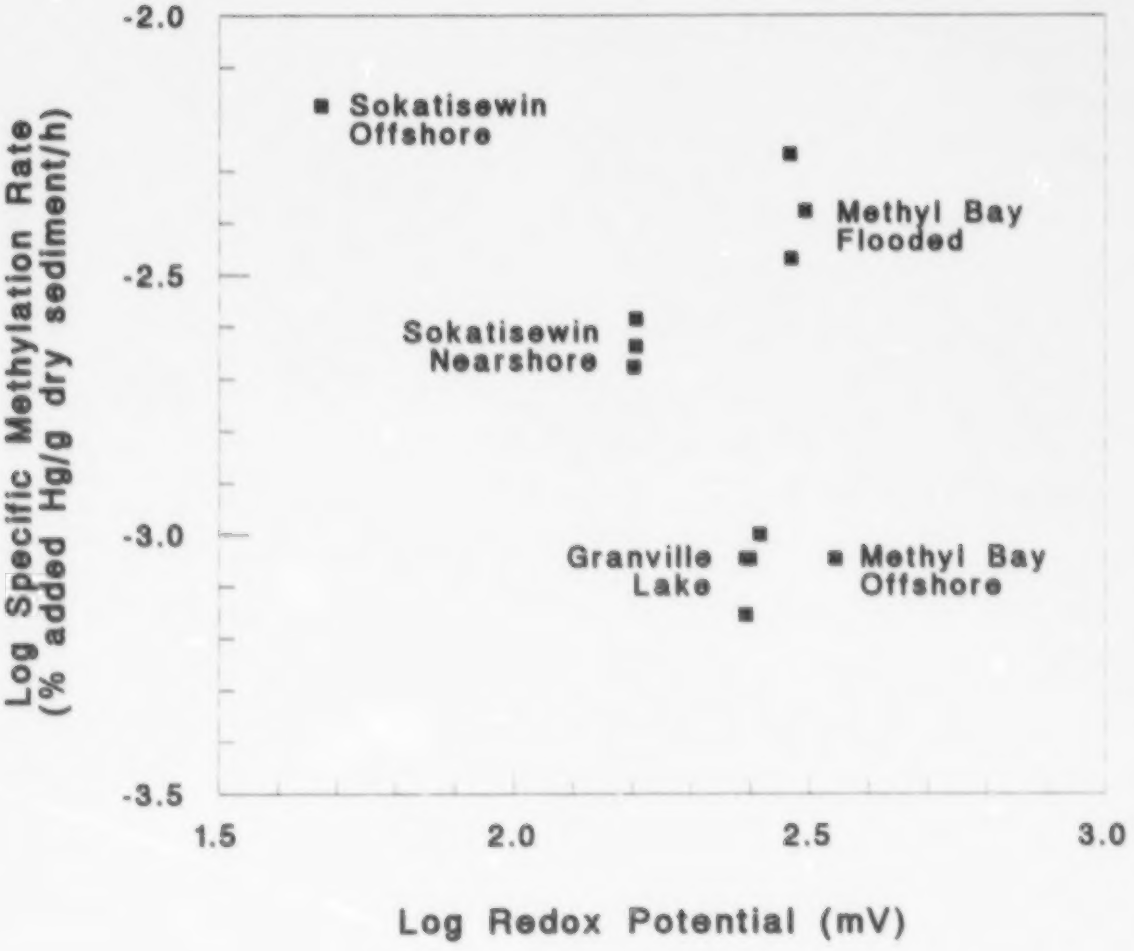

Figure 16. Relationship between the seasonal mean specific methylation rate and redox potential in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. Data from Ramsey (1989a).

increase as oxygen levels decline, provided there is sufficient organic substrate. Although these groups may be more active in reducing conditions, total microbial activity may decrease with decreasing redox potential because the energy yield of the anaerobic process is lower (Mitchell 1974). Thus, the large quantity of organic matter in flooded sediments influences mercury methylation in two ways; providing more bacterial substrate, and promoting the development of conditions which favour the types of bacteria capable of methylation.

3.2 Demethylation

The factors affecting microbial demethylation rates are less well-understood. In the 1986 and 1987 FEMP studies, demethylation rates were positively correlated with methylation rates (Figure 17) suggesting a common limiting factor. Organic matter availability was identified as the controlling variable (Ramsey 1987a and 1988a). However, there was no correlation between demethylation and methylation rates in 1988 nor were demethylation rates correlated with organic carbon (Figures 17 and 18). Instead, demethylation rates were positively correlated with sediment total mercury, microbial activity, and redox potential, and were negatively correlated with sediment pH (Ramsey 1989a). Total mercury was collinear with pH and both were eliminated as regulating variables based on the small range in pH among the stations (6.34-6.80), knowledge of the range of pH which can affect demethylation, and the collinearity of the variables (Ramsey 1989a). Microbial

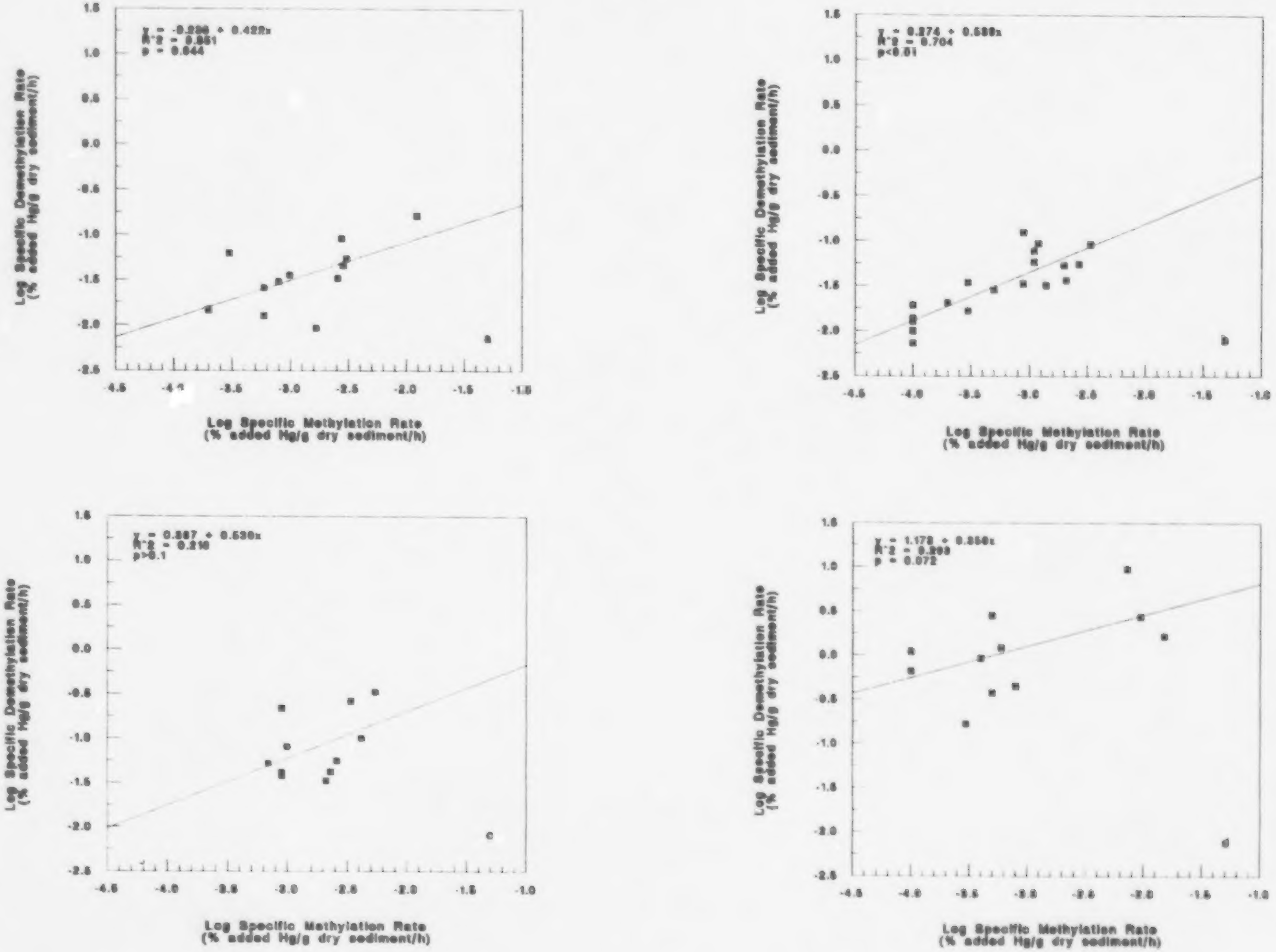

Figure 17. Relationship between seasonal mean specific rates of methylation and demethylation in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data from Ramsey (1987a, 1988a, 1989a, and 1990).

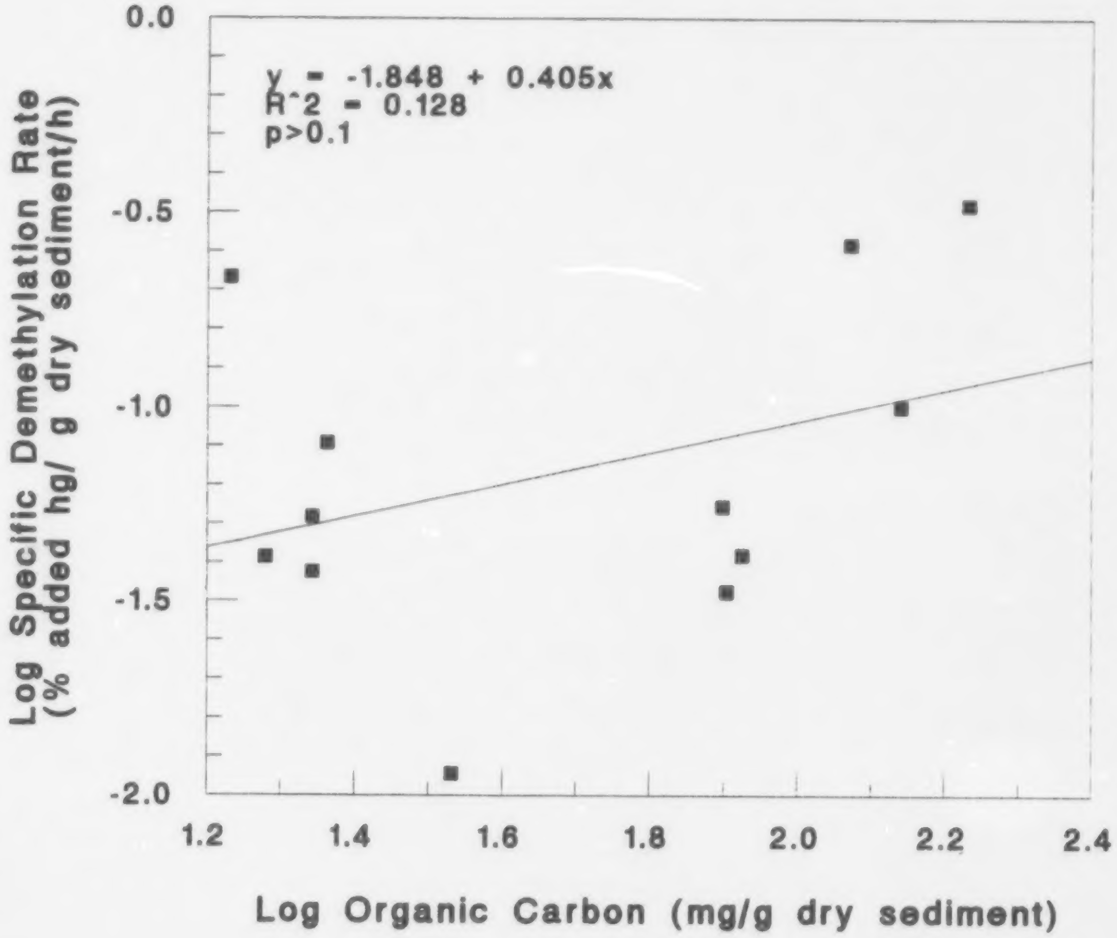

Figure 18. Relationship between seasonal mean specific demethylation rates and organic carbon concentrations in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. Data from Ramsey (1989a).

activity and redox also were collinear, suggesting it was only one of these factors which was responsible. There was no basis for differentiating between microbial activity and redox potential as the regulating factor, nor was there any obvious mechanism by which either could control demethylation (Ramsey 1989a).

A marginally significant correlation between methylation and demethylation rates was found in the 1989 survey of Methyl Bay, Stephens Lake, and Granville Lake (Figure 17). Demethylation also was positively correlated with sediment organic carbon, total nitrogen, and redox potential, and negatively correlated with sediment pH (Ramsey 1990). Again, pH was ruled out on the basis of other studies which examined the effect of pH on demethylation. Redox potential was ruled out as well. Although the differences in redox potential among the sites were highly significant, they were very small in absolute terms (range 149-168 mV). This left organic carbon and total nitrogen which, as was the case in the previous FEMP surveys (Ramsey 1987a, 1988a, and 1989a), also were intercorrelated. This importance of organic matter suggested total microbial activity was the controlling factor but this could not be confirmed directly because the rate of DIC production could not be measured in Stephens Lake due to high carbonate concentrations in the sediments which interfered with the analysis (Ramsey 1990).

An important distinguishing factor between methylation and demethylation is the lack of an influence of redox potential on demethylation rates. This may provide a partial explanation for the differential stimulation of methylation over demethylation in flooded sediments although additional study on the role of sediment oxygen conditions in regulating demethylation is required.

4.0 Duration of the Fish Mercury Problem

When the mercury problem on the Churchill River diversion was first identified in 1979, such problems were thought to be short-lived, with a duration of about 5 years. This projection was found to be incorrect, at least for reservoirs in northern Canada. There was no indication of a decline having started by 1981, five years after the impoundment of SIL and 7 years after Notigi Reservoir began filling. One of the main objectives of the FEMP, therefore, has been to develop and refine projections of the duration of the mercury problem on the Churchill River diversion. This has been approached by:

- monitoring microbial rates of mercury methylation and demethylation at flooded and reference sites in the CRD area;
- monitoring mercury levels in fish from flooded lakes in northern Manitoba; and
- measuring methylation and demethylation rates in older reservoirs located in the northern boreal forest zone.

The Department of Fisheries and Oceans was responsible for the monitoring of fish mercury levels in Southern Indian and Issett lakes. A detailed report of this study is presented by Strange et al. (1991). Monitoring of fish mercury on the CRD and in lakes and reservoirs on the Nelson River was the responsibility of the Province of Manitoba, Department of Natural Resources, Fisheries Branch. Green (1990) documents the data from this study and Ramsey (1991) presents an analysis of temporal trends between 1983 and 1989. The methylation and demethylation studies were undertaken by Agassiz North Associates Limited for Environment Canada, and are reported in Ramsey (1987a, 1988a, 1989a, and 1990).

4.1 Methylation on the CRD

Monitoring of microbial methylation and demethylation on the CRD has concentrated on two sites, a small bay on Southern Indian Lake, called Methyl Bay, representative of flooded conditions, and a reference site on Granville Lake near Leaf Rapids, which has not been affected by flooding (Figure 1).

Specific methylation rates and M/D at the flooded stations in Methyl Bay have been significantly higher than in the nearshore zone of Granville Lake in each of the six years of sampling (Table 7). Specific demethylation rates in Methyl Bay also significantly exceeded rates in Granville Lake in all years but 1984. Not only does the methylation balance remain shifted in favour of methyl mercury production, but there is no evidence of a sustained downward trend in the stimulatory affect of the flooded materials. No consistent declines have occurred in any of methylation, demethylation, or M/D. There was the suggestion of a decline in methylation rates in flooded sediments relative to the reference

Table 7. Comparison of seasonal mean specific methylation and demethylation rates (% added Hg methylated or demethylated (g dry sediment)$^{-1}$ h^{-1}) and corresponding methylation balance (M/D) in the flooded zone of Methyl Bay and in the nearshore zone of Granville Lake: 1984-1989. Values in parentheses are standard deviations of the mean for years when two (1984) or three (1985-1989) stations were sampled. Where no standard deviation is listed, only one station was sampled. Data from Ramsey and Ramlal (1987a and b) and Ramsey (1987a, 1988a, 1989a, and 1990).

	1984	1985	1986	1987	1988	1989
Methylation						
Methyl Bay	0.034 (0.026)	0.022 (0.012)	0.012	0.002 (0.001)	0.0043 (0.0010)	0.0106 (0.0041)
Granville Lake	0.012	0.006 (0.005)	0.0008 (0.0002)	0.0002 (0.0001)	0.0009 (0.0002)	0.0005 (0.0001)
Ratio	2.8	3.7	15.0	10.0	4.8	21.2
Demethylation						
Methyl Bay	0.089 (0.040)	1.200 (0.133)	0.162	0.0468 (0.0127)	0.2312 (0.1165)	4.6102 (4.2504)
Granville Lake	0.162	0.669 (0.216)	0.031 (0.005)	0.0166 (0.0027)	0.0581 (0.0205)	2.2640 (0.9180)
Ratio	0.5	1.8	5.2	2.8	4.0	2.0
M/D						
Methyl Bay	0.383 (0.176)	0.018 (0.012)	0.077	0.0434 (0.0067)	0.0236 (0.0158)	0.0045 (0.0043)
Granville Lake	0.076	0.009 (0.009)	0.026 (0.003)	0.0102 (0.0069)	0.0159 (0.0159)	0.0003 (0.0002)
Ratio	5.0	2.0	3.0	4.3	1.5	15.0

site in 1987 and 1988, but this decline did not continue into 1989, when the largest differential in the 6 year study was recorded (Table 7).

4.2 Mercury in Fish from SIL, the CRD, and Nelson River

Mercury levels in northern pike, walleye, and whitefish in Rat, Threepoint, and Stephens lakes (Figure 1) were surveyed by Manitoba Natural Resources annually between 1983 and 1989. Elevated mercury levels developed in fish from Rat and Threepoint lakes after flooding by the Churchill River diversion between 1974 and 1976, and in Stephens Lake due to flooding by the Kettle Generating Station between 1971 and 1972. Mercury levels have peaked in all species in all three lakes, and appear to be starting to decline. This decline is expected to be slow, however, as indicated by mercury levels in Stephens Lake walleye and pike which remain 3 and 4 times higher than pre-development values 18 years after impoundment. The results of the Fisheries Branch study are summarized below for each lake and are detailed in Green (1990) and Ramsey (1991).

4.2.1 Rat Lake

Rat Lake was flooded between 1974 and 1976 as Notigi Reservoir was created in the Rat River valley (Figure 1). Lake level rose about 10 m (Newbury et al. 1984) and lake area increased approximately 248% (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.).

The whitefish mercury data record for the lake starts in 1980, with the peak level, 0.25 $\mu g\ g^{-1}$, occurring in 1984 (Figure 19). Whitefish mercury was stable between 1986 and 1989 at 0.2 $\mu g\ g^{-1}$. Thus, 14 years after the start of impoundment, whitefish mercury remains more than a factor of 2 higher than the average for northern Manitoba lakes remote from hydro development (<0.1 $\mu g\ g^{-1}$; Derksen and Green 1987) and 2.5 to 4 times the pre-development levels upstream in SIL (0.05-0.07 $\mu g\ g^{-1}$; Bodaly et al. 1984) and downstream in Wuskwatim Lake (0.08 $\mu g\ g^{-1}$ in 1970; Bodaly et al. 1984).

The walleye mercury data record for Rat Lake starts in 1978, with uniformly high levels occurring between 1978 and 1984 (Figure 19). Although a decline appears to have occurred between 1978 and 1983, its significance cannot be tested as the pre- and post-1983 data are not directly comparable. The pre-1983 data were not standardized to length in the same manner as the later data. The early samples also were biased to the larger (300+ mm) individuals in the population (Green 1986), which probably explains the apparently higher mean levels during this period.

There was a significant decline in walleye mercury between the 1984 and 1985 samples and again between 1988 and 1989 (Figure 19). Nevertheless, 14 years after impoundment, walleye mercury in Rat Lake remained more than 3 times higher than both the average for northern Manitoba lakes remote from hydro development (0.3-0.4 $\mu g\ g^{-1}$; Derksen and Green 1987) and the pre-development levels upstream in SIL (1971-1975, 0.19-0.30 $\mu g\ g^{-1}$; Bodaly et al. 1984) and

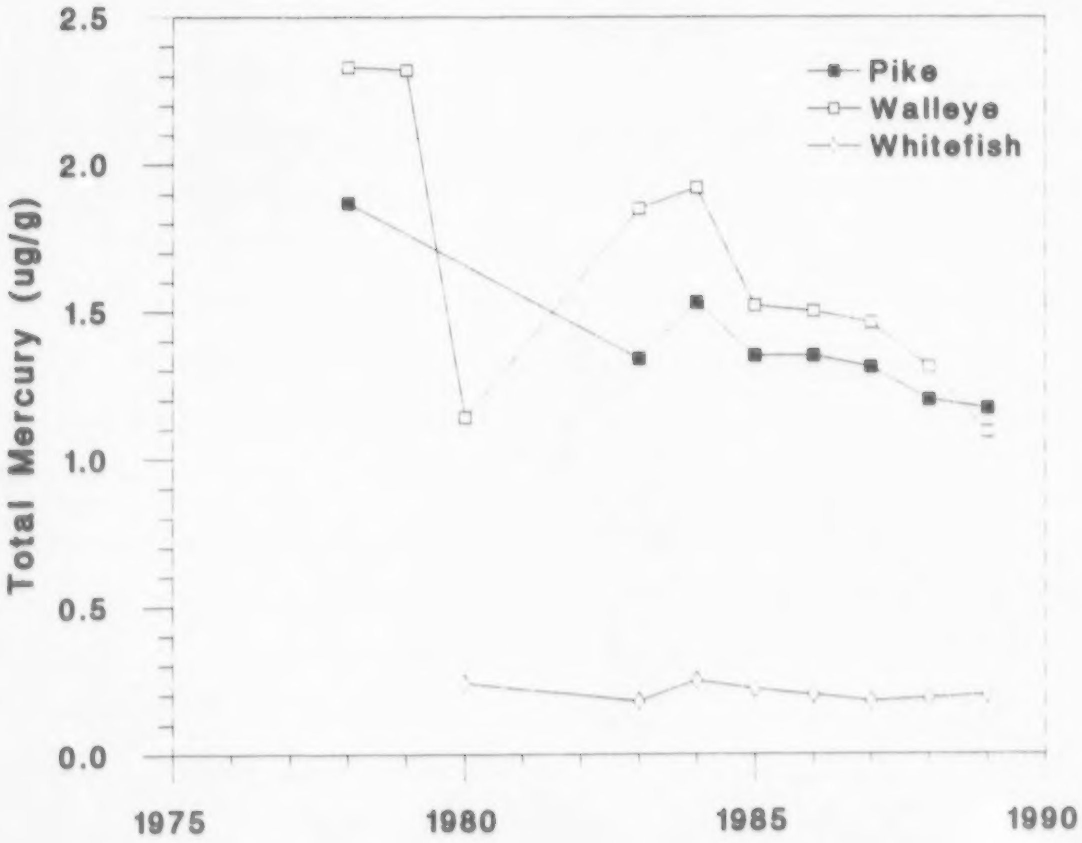

Figure 19. Mean total mercury concentrations in northern pike, walleye, and whitefish from Rat Lake, 1978-1989. Pre-1983 data from Green (1986). Later data are length-standardized means from Green (1990).

downstream in Wuskwatim Lake (1970-1975, 0.25-0.40 $\mu g\ g^{-1}$; Bodaly et al. 1984).

The northern pike mercury data record also starts in 1978 (Figure 19). Again, the significance of any decline between 1978 and 1983 cannot be tested. Data for the two periods are not directly comparable, as described above for walleye. There was a small (0.15 $\mu g\ g^{-1}$) but significant decline between 1986 and 1988, with the lower level also recorded in 1989. As in the other species, pike mercury levels remain substantially elevated 14 years after impoundment. In 1989, the mean pike mercury level in Rat Lake was a factor of 3 higher than the average for northern Manitoba lakes remote from hydro development (0.3-0.4 $\mu g\ g^{-1}$; Derksen and Green 1987) and was more than 3.5 times the pre-development level upstream in SIL (1971-1973, 0.26-0.32 $\mu g\ g^{-1}$; Bodaly et al. 1984).

4.2.2 Threepoint Lake

Threepoint Lake was flooded by the Churchill River diversion in 1976 (Figure 1). Lake level rose about 3.9 m and area increased 39% (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.).

The post-impoundment peak in whitefish mercury occurred sometime between 1976 and 1981 (Figure 20). The timing and magnitude of the peak is unknown due to the lack of samples during this period. The level was constant, at about 0.2 $\mu g\ g^{-1}$, between 1983 and 1989. Although there are no pre-development whitefish mercury data for Threepoint Lake, the current level most probably

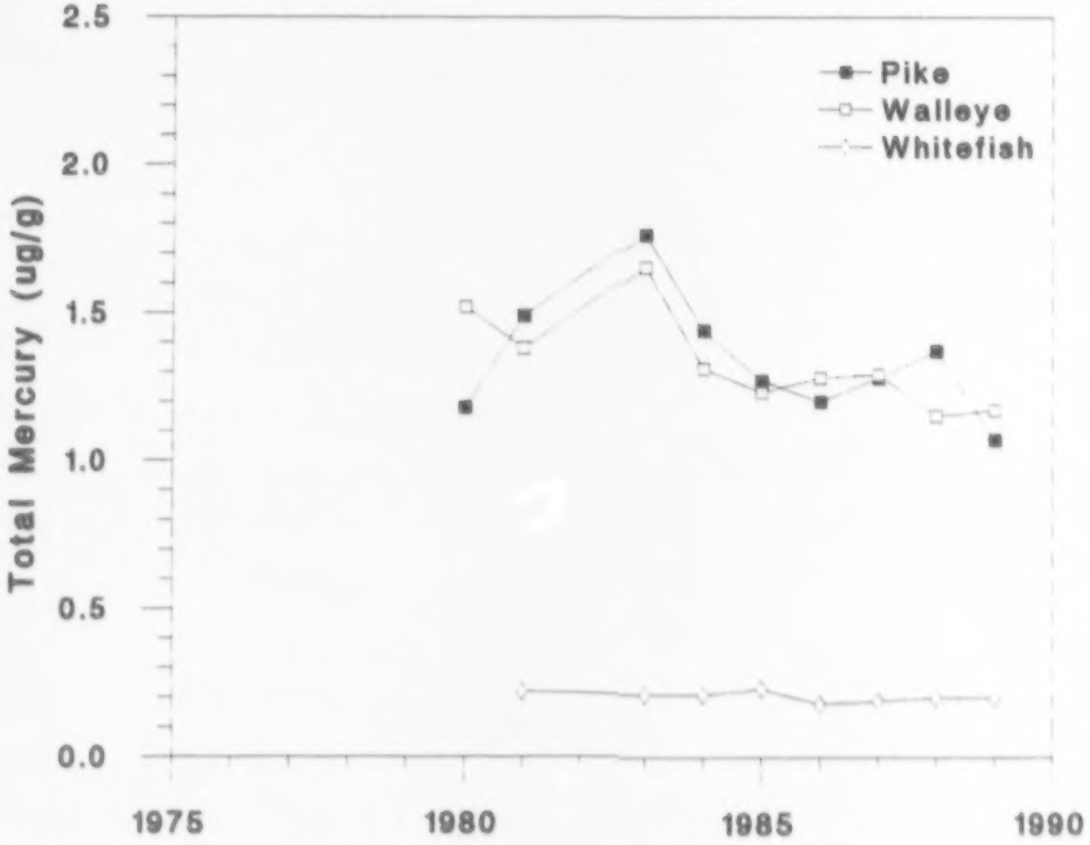

Figure 20. Mean total mercury concentrations in northern pike, walleye, and whitefish from Threepoint Lake, 1980-1989. Pre-1983 data from Green (1986). Later data are length-standardized means from Green (1990).

is elevated, as it is a factor of 2 higher than the average for northern Manitoba lakes remote from hydro development (<0.1 μg g^{-1}; Derksen and Green 1987) and 2.5 to 4 times the pre-development levels upstream in SIL (0.05-0.07 μg g^{-1}; Bodaly et al. 1984) and downstream in Wuskwatim Lake (0.08 μg g^{-1} in 1970; Bodaly et al. 1984).

The post-impoundment peak in walleye mercury, 1.64 μg g^{-1}, appears to have occurred in 1983 (Figure 20). The mean level decreased significantly between 1983 and 1984, to 1.3 μg g^{-1}, and remained at this concentration through 1989. Twelve years after impoundment and Churchill River diversion, walleye mercury in Threepoint Lake was more than a factor of 3 higher than both the average for northern Manitoba lakes remote from hydro development (0.3-0.4 μg g^{-1}; Derksen and Green 1987) and the pre-development concentrations upstream in SIL (1971-1975, 0.19-0.30 μg g^{-1}; Bodaly et al. 1984) and downstream in Wuskwatim Lake (1970-1975, 0.25-0.40 μg g^{-1}; Bodaly et al. 1984).

The post-impoundment peak in pike mercury, 1.75 μg g^{-1}, also appears to have occurred in 1983 (Figure 20). The mean level declined significantly over the following 3 years, reaching 1.2 μg g^{-1} in 1986. No further significant declines occurred between 1986 and 1989.

4.2.3 Stephens Lake

Stephens Lake was created in two stages in the autumns of 1970 and 1971 by the construction of the Kettle Generating Station on the Nelson River at Gillam, Manitoba (Water Resources Branch 1974). The depth of flooding at the dam is 30 m and total water area increased by about 300% (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.)

The trend of slowly declining fish mercury levels after impoundment is evident in Stephens Lake pike and walleye but not in whitefish. There was no significant change in whitefish mercury between 1983 and 1989 (Figure 21). Averaging 0.16 $\mu g\ g^{-1}$ during this period, whitefish mercury in Stephens Lake was over 60% higher than the average for northern Manitoba lakes remote from hydro development (<0.1 $\mu g\ g^{-1}$; Derksen and Green 1987) and was double the mean level upstream in Split Lake during the same period (Ramsey 1991). Although there are no pre-development whitefish mercury data for Stephens Lake, the current levels clearly appear to be elevated, consistent with the demonstrably elevated levels in other species in the lake.

Walleye mercury in Stephens Lake declined significantly, from 1.37 $\mu g\ g^{-1}$ in 1983 to 0.81 $\mu g\ g^{-1}$ in 1984, and stabilized at this lower level from 1984 to 1989 (Figure 21). The timing and magnitude of the peak post-impoundment walleye mercury level will never be known with certainty because no samples were collected in the first 10 years after flooding. It is difficult to determine if the decline between 1983 and 1984 represented a real decrease or was an

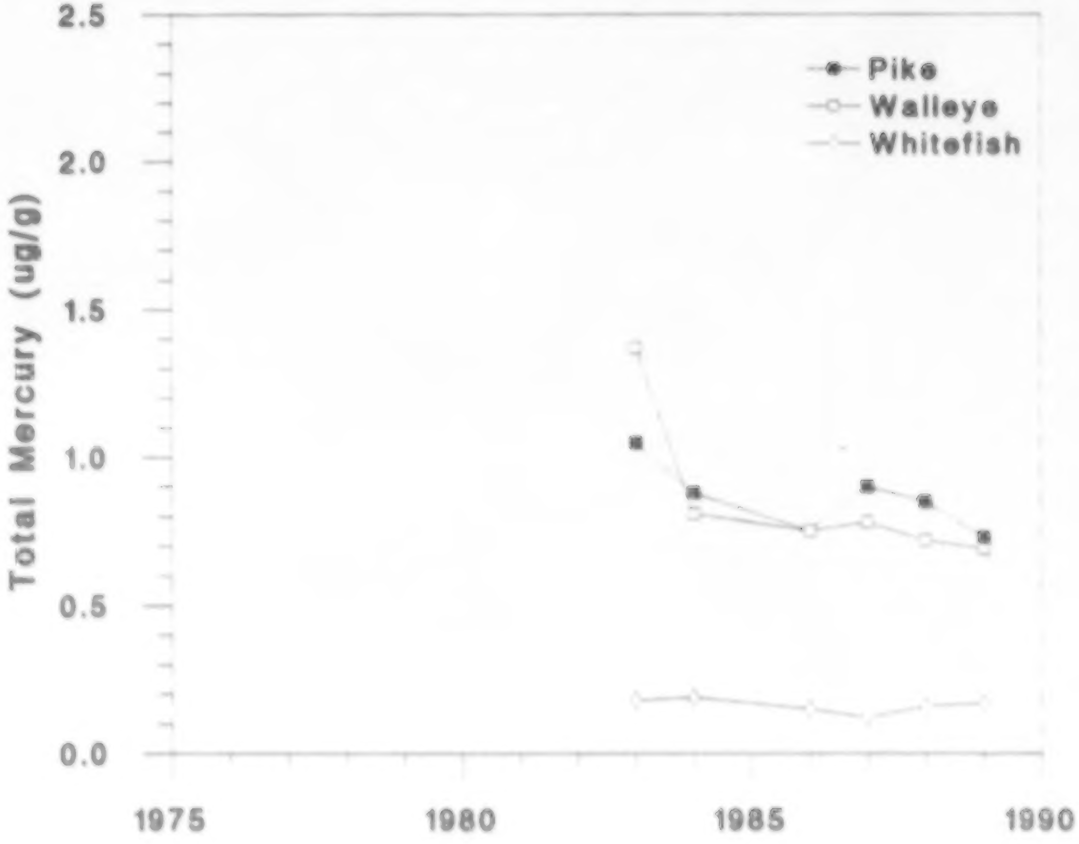

Figure 21. Mean total mercury concentrations in northern pike, walleye, and whitefish from Stephens Lake, 1983-1989. All data are length-standardized means from Green (1990).

artefact of a small (n=9), size-biased sample in 1983. As late as 1989, 18 years after impoundment, walleye mercury was 3 times higher than the pre-development level of 0.25 μg g^{-1} reported for Moose Lake (n=1, Department of Fisheries and Oceans, Southern Operations Directorate, Winnipeg, MB), which was inundated in the formation of Stephens Lake.

Pike mercury also declined significantly, from 1.05 to 0.75 μg g^{-1}, between 1983 and 1986, and remained at this lower level through 1989 (Figure 21). The timing of the peak in pike mercury also will never be certain due to the lack of data for the 1971-1982 period. The standardized pike mercury level in 1989 was 4 times the pre-development concentration of 0.20 μg g^{-1} reported for Moose Lake (n=4, Department of Fisheries and Oceans, Southern Operations Directorate, Winnipeg, MB).

4.2.4 Southern Indian Lake

The level of Southern Indian Lake was raised 3 m in August, 1976, to facilitate diversion of the Churchill River into the Rat-Burntwood and Nelson river systems at South Bay. Lake area increased approximately 9% due to impoundment (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.)

Fish mercury levels have been monitored at a number of sites on SIL regularly between 1978 and 1988. This section will concentrate on the data for three species, northern pike, walleye, and whitefish, from four sites, Area 5, Area 4, the Channel, and Camp 9 (Figure 22). Data for other species or areas of the lake may be

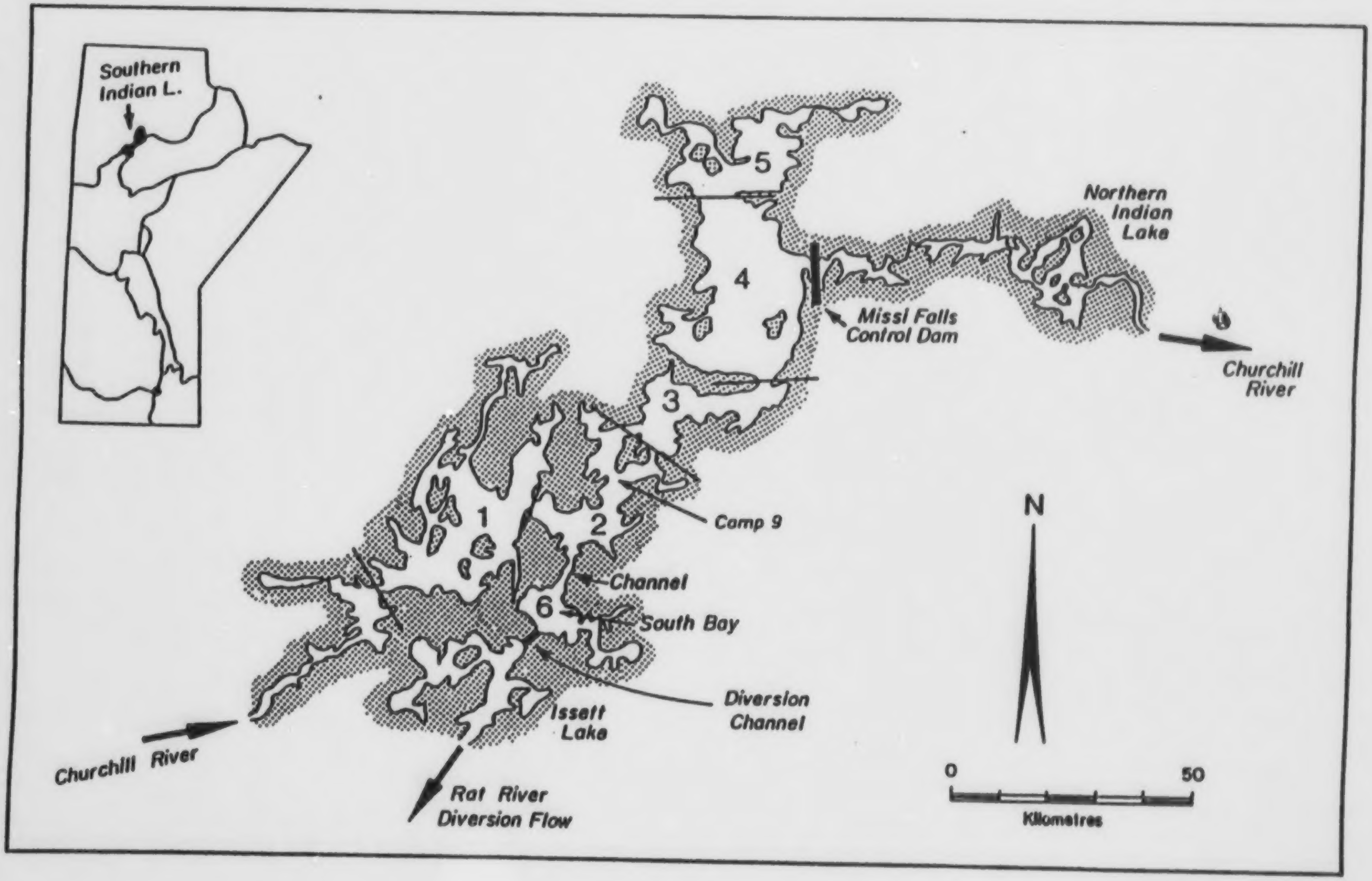
Southern
Indian L.
5
4
3
2
1
6
Northern
Indian
Lake
Missi Falls
Control Dam
Churchill
River
Camp 9
Channel
South Bay
Diversion
Channel
Issett
Lake
Churchill River
Rat River
Diversion Flow
N
0
50
Kilometres

Figure 22. Location of Department of Fisheries and Oceans fish mercury sampling locations on Southern Indian Lake. From Strange et al. (1991).

found in Strange et al. (1991).

Mercury levels in all three species increased sharply after impoundment in 1976 (Figures 23, 24, and 25). Whitefish mercury peaked between 1978 and 1981, depending on location. Levels have since declined and, by 1988, had returned to near pre-impoundment concentrations in all surveyed regions with the exception of Area 5. Walleye mercury peaked between 1979 and 1981 (Figure 24). Levels have since declined from these peak values but, as late as 1988, mercury levels were still well above the pre-impoundment concentration. In contrast to the whitefish and walleye, the mean mercury concentration in northern pike has not yet started to decline at any site and may be continuing to increase (Figure 25).

4.2.5 Comparison of Temporal Trends in Fish Mercury

There are several differences in the temporal trends in fish mercury levels in SIL compared with sites on the CRD and Nelson River. The first difference is the clearly declining levels in SIL whitefish, in contrast to the other 3 lakes where no substantial or sustained decreases have occurred. Secondly, the decline in walleye mercury levels in SIL also has been larger in proportion to the initial increase than at the other sites, following the trend in whitefish mercury. Another difference is the occurrence of lower mercury levels in walleye than in pike from SIL, unlike the other sites. Finally, pike mercury in SIL has not declined at all, contrary to the trends in other species from SIL and to the slight downward trends in pike mercury at the other sites.

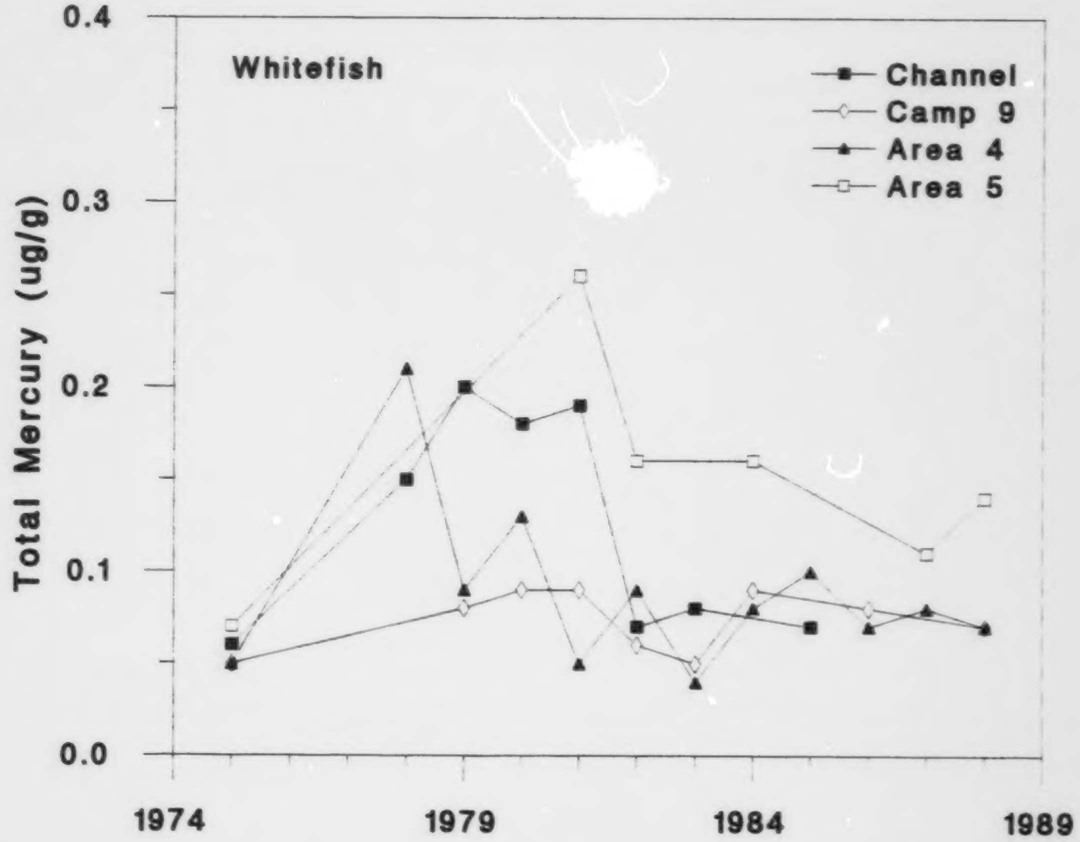

Figure 23 Mean length-standardized mercury concentrations in whitefish from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991).

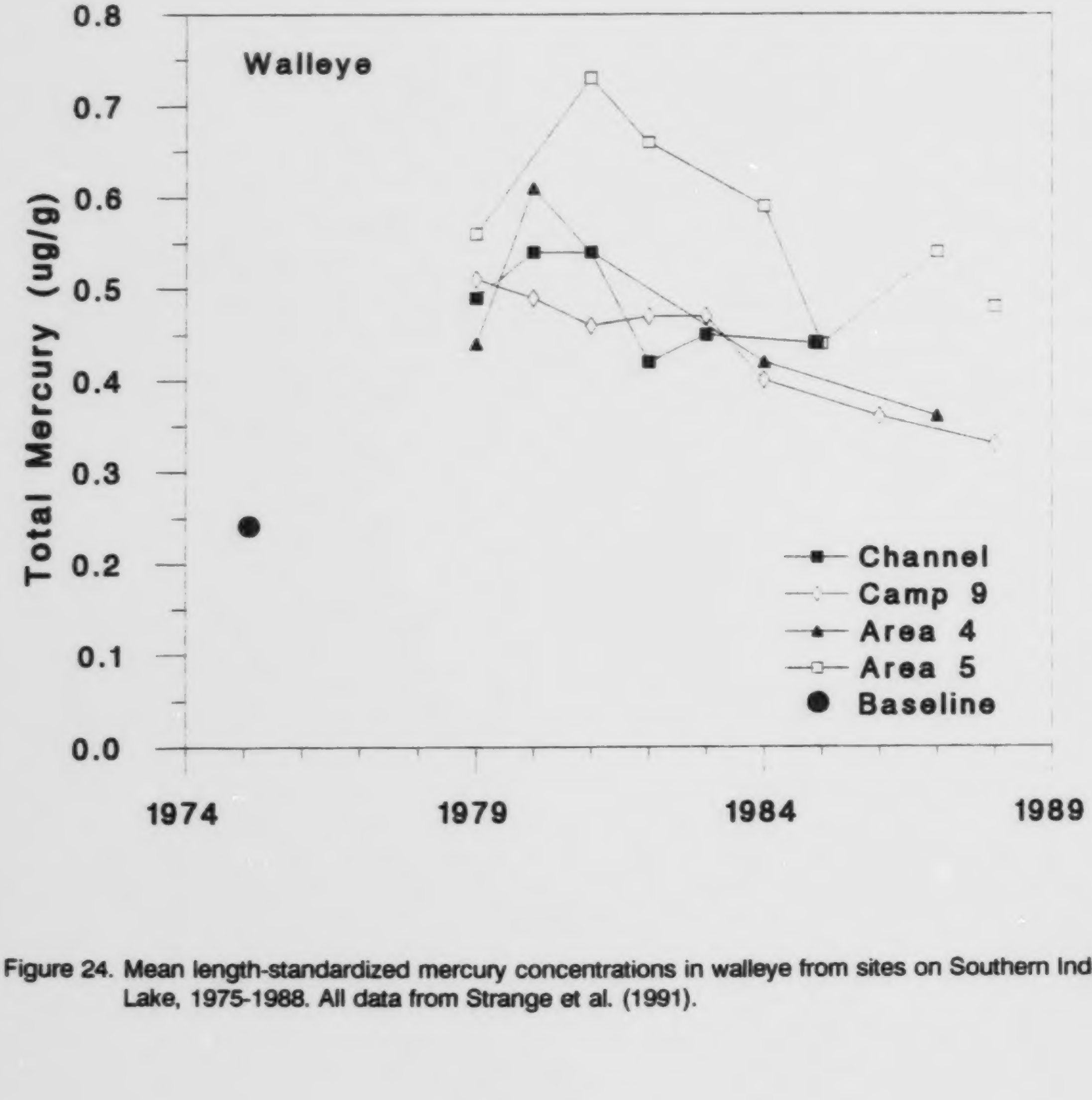

Figure 24. Mean length-standardized mercury concentrations in walleye from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991).

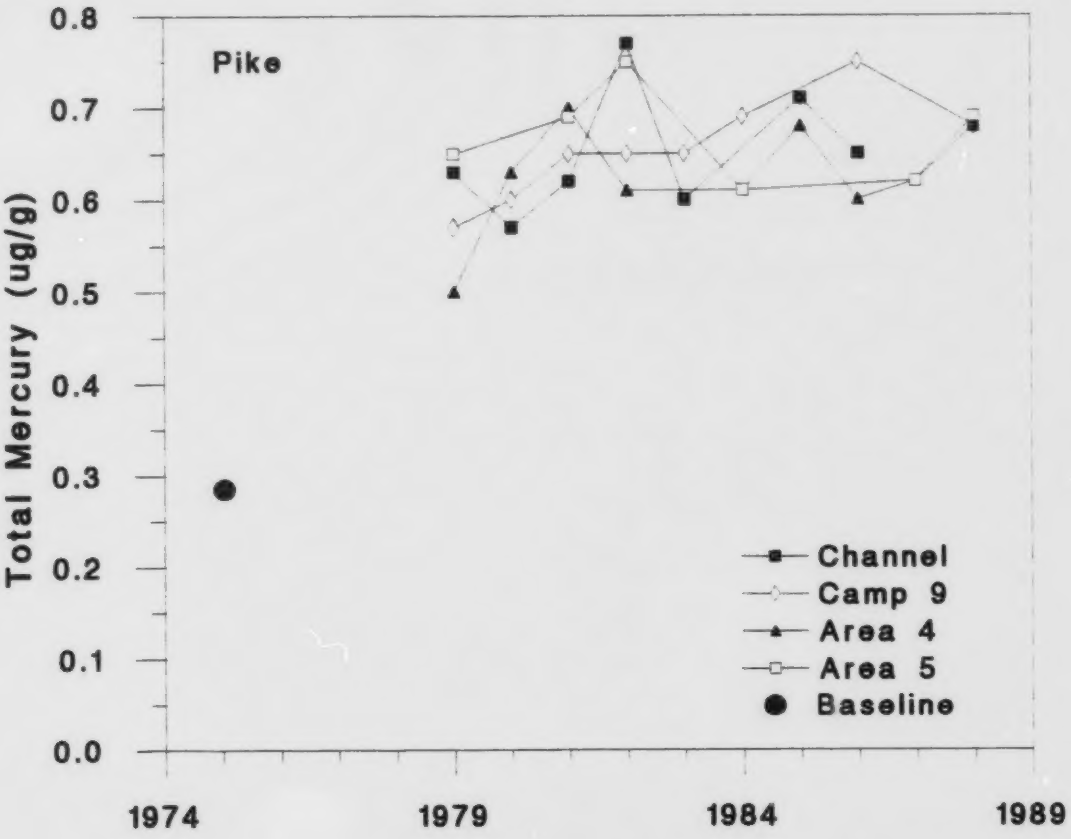

Figure 25. Mean length-standardized mercury concentrations in northern pike from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991).

These differences in temporal trends can be explained by two factors; (i) interaction of the species' different habitat preferences with the relatively small extent of flooding on SIL compared to other flooded lakes on the CRD and Nelson River and, (ii) the changes in limnological conditions which have occurred in Notigi Reservoir and Stephens Lake.

Pike prefer the nearshore zone (Inskip 1982), spending most of their time over the flooded terrain which is the site of elevated methyl mercury production. Net methylation in the flooded zone of SIL remains elevated, with no evidence of a decline since 1984 (see Section 4.1), and mercury levels in pike reflect this continued exposure. Given that the same types of materials were flooded in Notigi Reservoir, and these materials have been submerged for about the same length of time as in SIL, there is no reason to expect the ability of the flooded materials to stimulate methyl mercury production has decreased. Nevertheless, mercury levels in Rat Lake pike have declined. There must be some other explanation and a clue is provided by the timing of the decreased pike mercury levels in Rat Lake which coincided with the occurrence of elevated suspended sediment levels in Notigi Reservoir (McCullough 1990). Nearshore sedimentation is a consequence of increased shoreline erosion, as seen in SIL (Newbury and McCullough 1984; McCullough 1990). Sediment deposition reduces the area of exposed organic matter and, therefore, the area of reservoir sediments supporting elevated net microbial methylation, resulting in reduced methyl mercury availability and a lower mercury level in pike. The covering of organic matter with inorganic sediment appears to effectively prevent the stimulation of microbial methylation (Hecky et al. 1987a). Consistent with this hypothesis, a similar increase in turbidity, indicating an increase in shoreline erosion, has been noted in Stephens Lake (Berger and Ramsey 1991). The small declines in walleye mercury levels in Rat and Stephens lakes are consistent with the declines in pike mercury and are probably due to the same cause.

In contrast to pike, adult whitefish are most often found mid-lake in deeper, cooler water. The whitefish only move into the nearshore areas in the late autumn when water temperatures drop below 10 °C (Derksen and Green 1987). In SIL, where only about 10% of the lake is comprised of flooded terrain (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.), this habitat preference places the whitefish over pre-existing lake sediments in the summer during the period of peak methyl mercury production. Net methylation in these pre-existing lake sediments has been at baseline levels since at least 1984 (Ramsey 1991). Methyl mercury produced in the flooded zone also appears to remain there, at least in Methyl Bay (Ramsey 1991). Thus, whitefish in much of SIL probably are not exposed to elevated methyl mercury levels during their daily activities.

This explanation accounts for the current low levels of mercury in SIL whitefish, but it does not account for the elevated levels which occurred immediately after impoundment. Either methyl mercury was exported offshore during this period or, more likely, the initial period of active shoreline erosion after impoundment resulted in organic matter being carried offshore where it could stimulate microbial methylation. Over time, eroding inorganic material would bury these offshore organics, reducing offshore methylation and, consequently, mercury levels in whitefish. In support of this hypothesis, Newbury and McCullough (1984) found organic matter of terrestrial origin as far as 300 m offshore (the maximum distance offshore that was sampled) and 250 m beyond the pre-existing lake shoreline. The area of exposed organic matter capable of stimulating microbial methylation clearly was much larger in the past than at present, and this explains the higher whitefish mercury levels in SIL immediately after impoundment. At the more extensively flooded sites such as Rat and Stephens lakes, where lake area occurs over flooded terrain, whitefish behaviour would not exclude them from exposure to flooded areas and the elevated methyl mercury levels found there. As a result, whitefish mercury levels in these lakes are higher than in SIL and decline more slowly.

Walleye are wide-ranging, moving into littoral areas at night and retreating to the deeper areas of the lake during the day (Derksen and Green 1987). This behaviour should reduce the exposure of walleye to the elevated methyl mercury levels in the flooded zone of SIL relative to the pike, accounting for the lower mercury levels in SIL walleye. In the more extensively flooded lakes, walleye behaviour does not provide an escape from flooded terrain, resulting in the similar mercury levels in pike and walleye from

Rat and Stephens lakes. The considerable decline in walleye mercury in SIL since 1980-81 can be accounted for in the same way that the decline in whitefish mercury is explained.

The fish mercury data for lakes on the CRD and Nelson River indicate declines are occurring at some but not all locations and are proceeding in a slow and discontinuous manner. Because the declines are slow and erratic in time and space, and the data record for all lakes is short relative to the rate of decline, it is difficult to base predictions of duration on the observed declines in fish mercury levels.

4.3 Methylation in Older Reservoirs

In addition to the annual measurements of microbial methylation and demethylation rates in SIL and Granville Lake, several older reservoirs were surveyed between 1986 and 1989. Reservoirs sampled in 1986 included the Ogoki Reservoir in northwestern Ontario, created in 1943, and Laurie Reservoir #2 situated on the Laurie River, flooded in the early 1950's (Figure 1). Sipiwesk Lake on the Nelson River, flooded in 1960 by the Kelsey Dam, was sampled in 1987. Sokatisewin Lake on the Churchill River in Saskatchewan, created in 1929 by the Island Falls Generating Station, was sampled in 1988.

A comparison of methylation and demethylation rates in the flooded zones of the SIL, Ogoki, and Laurie reservoirs in 1986 suggested that rates of both processes declined over time according to negative exponential models (Ramsey 1987a). These models were originally formulated using an analytical approach. They have been reformulated as regression models in this report for comparison with later versions (Figures 26, 27, and 28). Curve shapes differ slightly from those originally presented, but the estimates of recovery time are essentially the same. Use of a regression approach permitted the mathematic description of the decline in methylation balance over time, with a second order polynomial providing the best fit.

Based only on the 1986 data, methylation rates were projected to return to the baseline level measured in Granville Lake 57 years after impoundment (Figure 26). Demethylation rates were estimated to return to baseline 44 years after impoundment (Figure 27) with the methylation balance returning to baseline after 53 years (Figure 28). These models were tested and modified over the following three years with additional data from Methyl Bay and the other older reservoirs.

Even with the additional 3 years of measurements, the models describing the evolution of methylation and demethylation rates and the methylation balance over time remain very similar to those based only on the 1986 survey data. The negative exponential equation has been retained for the methylation and demethylation models (Figures 29 and 30) because it consistently provided the best fit to the data. This model also is consistent with the cause of the reservoir mercury problem. The rate of decomposition of organic matter also follows a negative exponential model (Hodkinson 1975; Anderson et al. 1978). The estimated time required for the

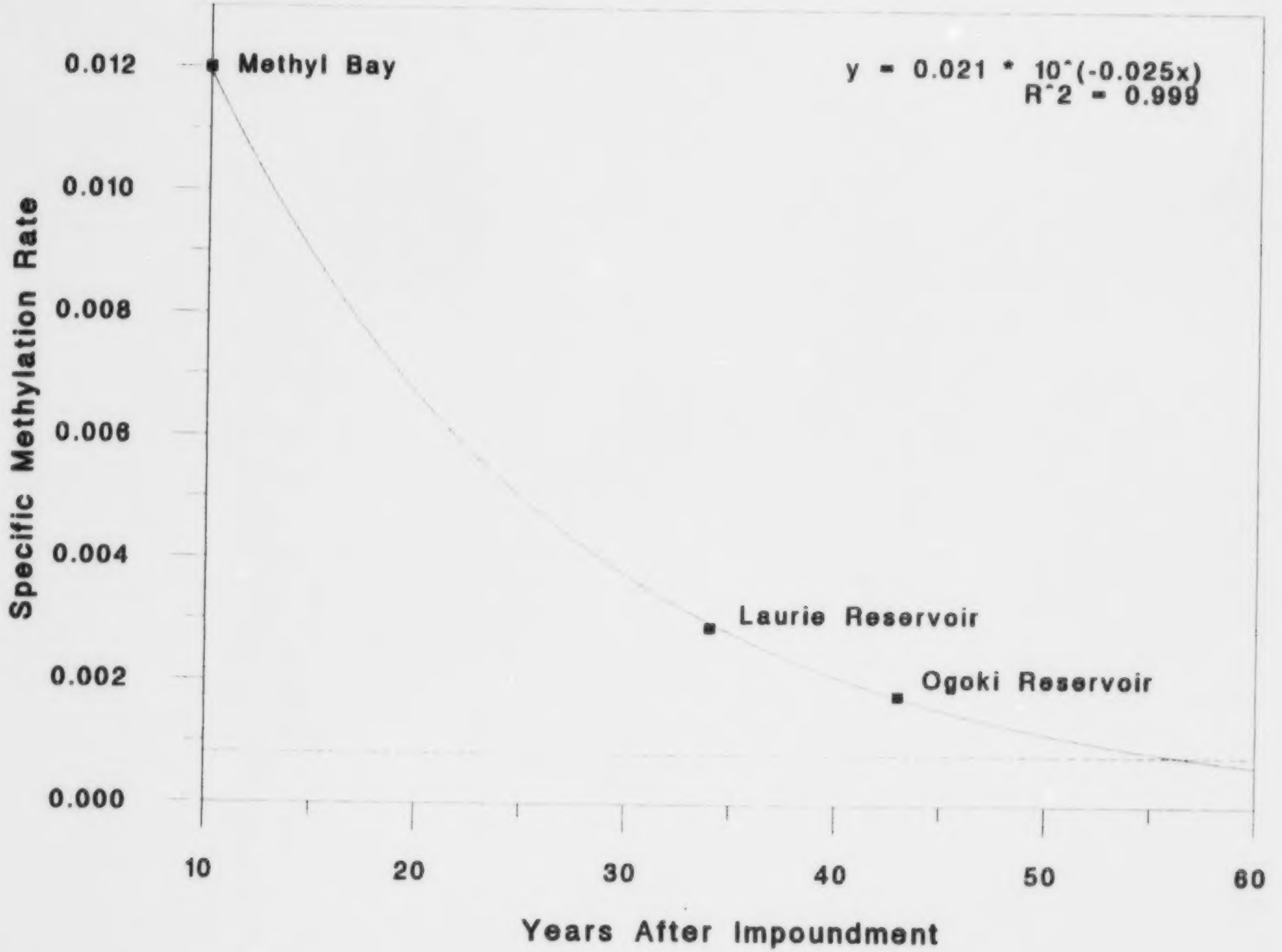

Figure 26. Relationship between the seasonal mean specific methylation rate (% added Hg methylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b).

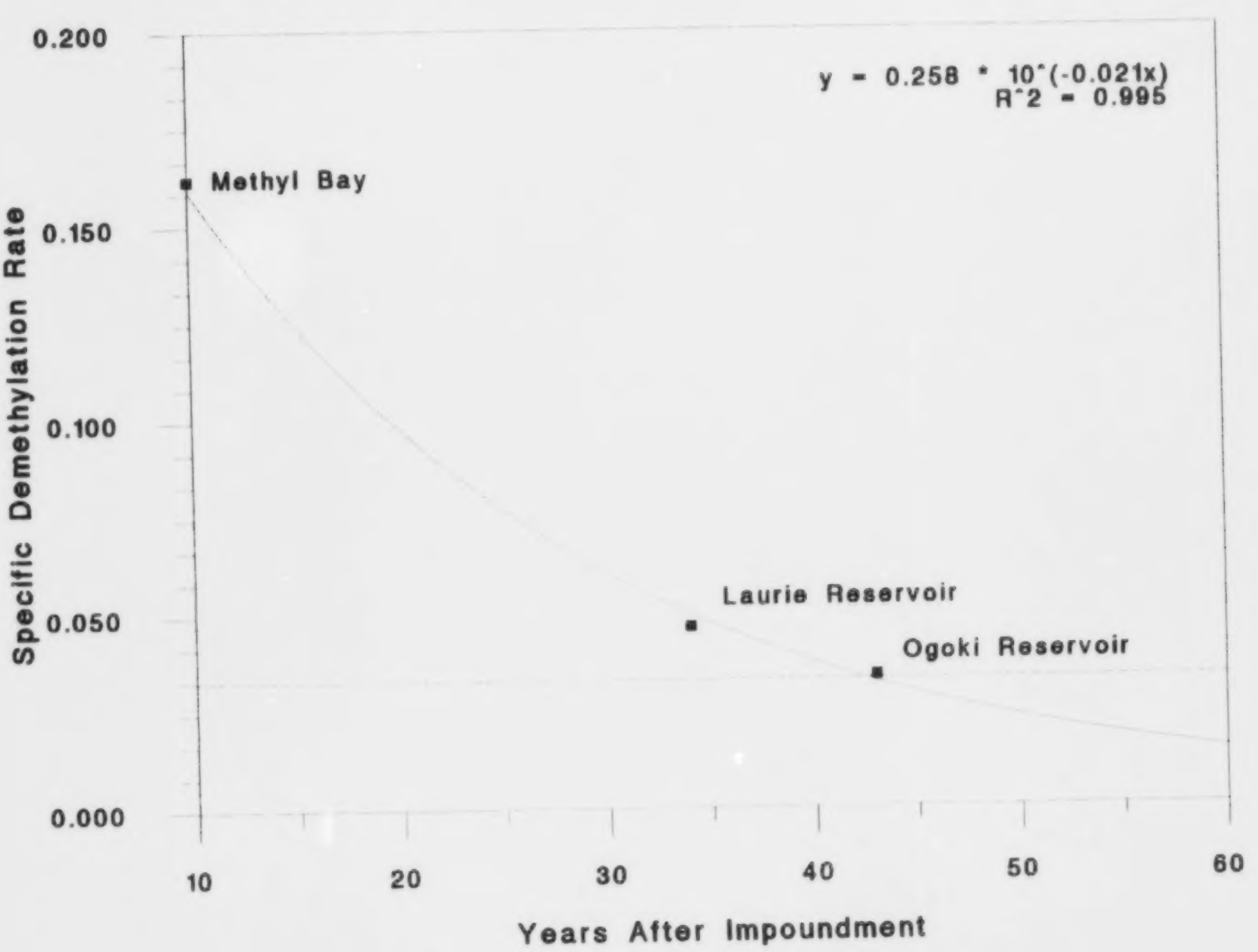

Figure 27. Relationship between the seasonal mean specific demethylation rate (% added Hg demethylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b).

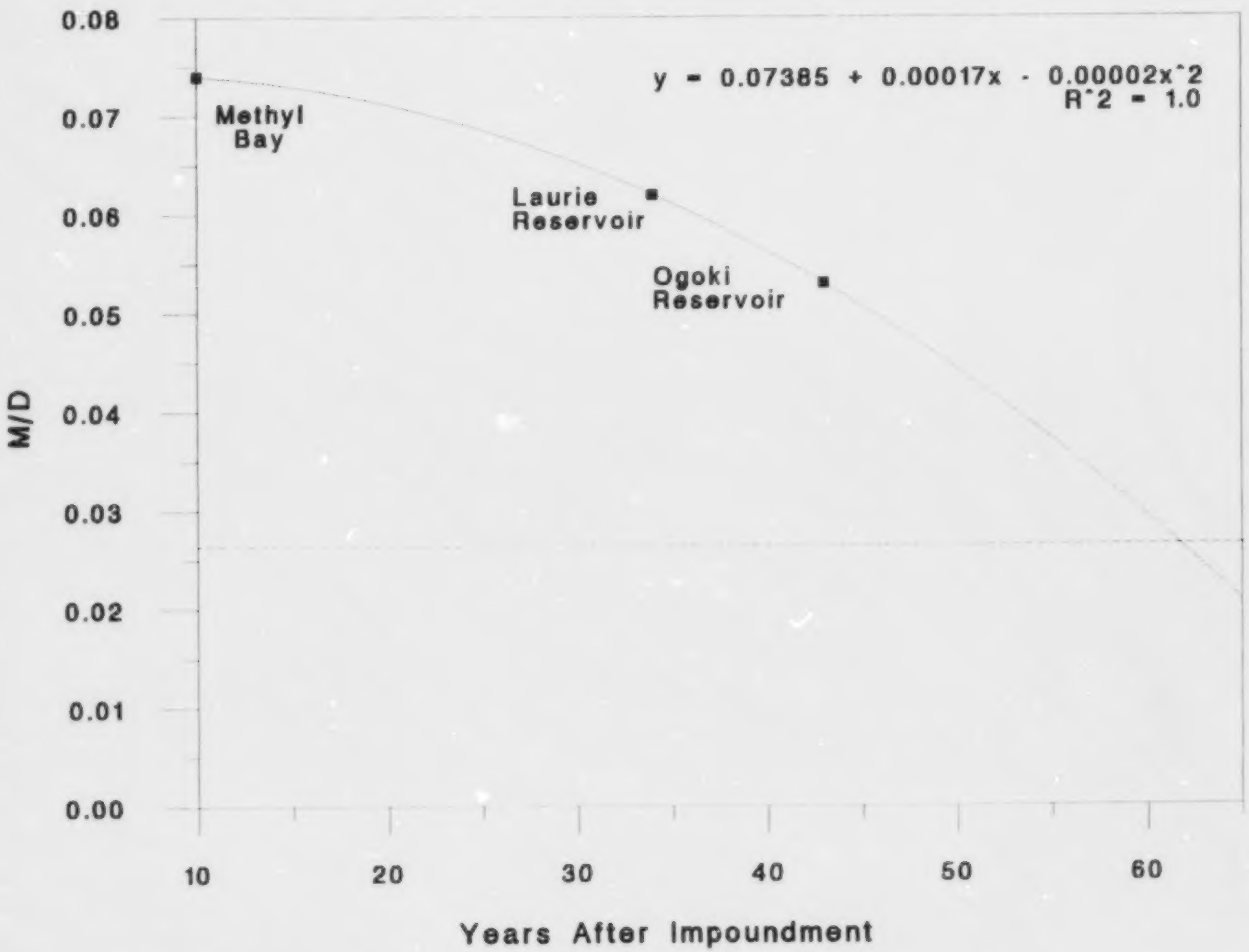

Figure 28. Relationship between the seasonal mean methylation balance (M/D) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b).

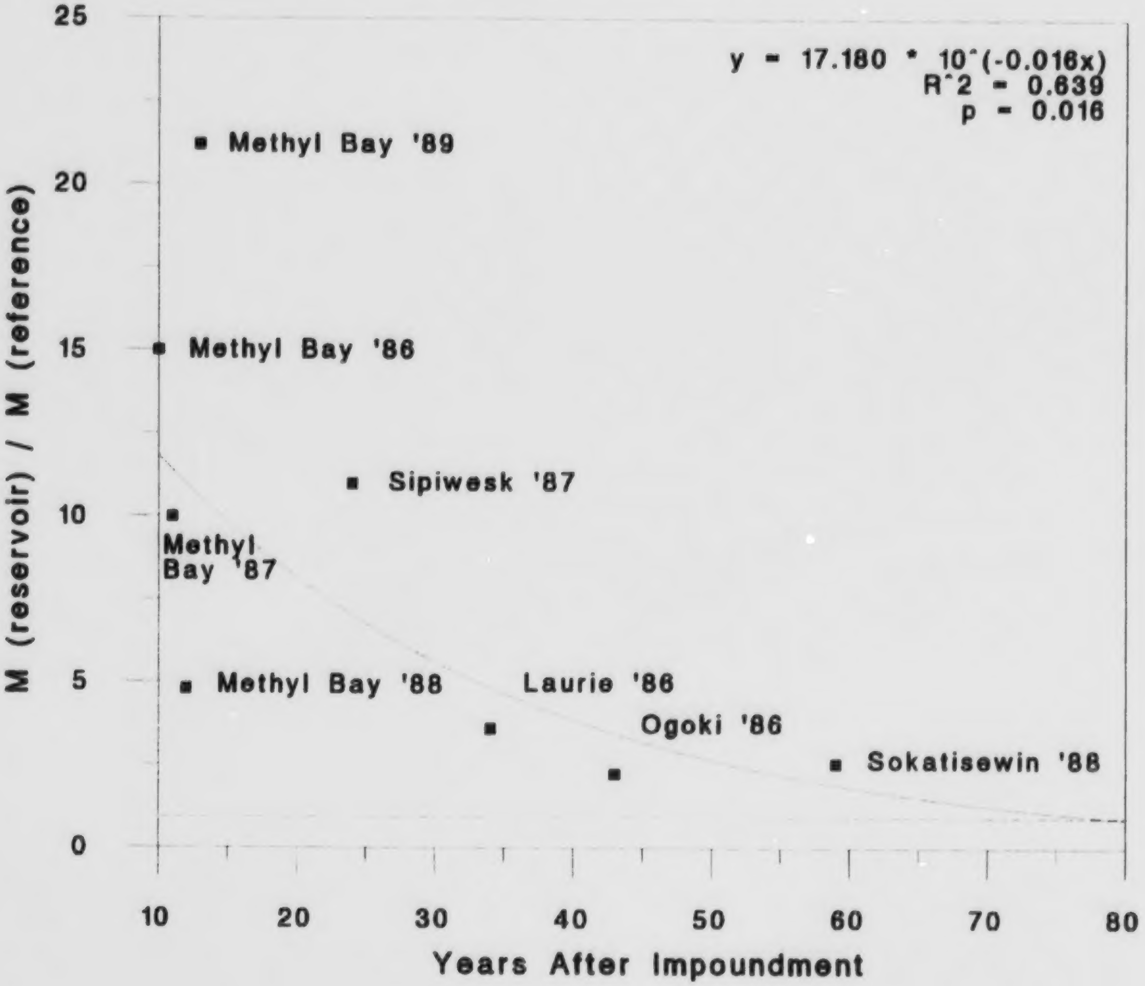

Figure 29. Relationship between the elevation in seasonal mean specific methylation rate (% added Hg methylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed between 1986 and 1989. Dashed line represents equality of rates in reservoir and reference lake sediments. Granville Lake was the reference site in all years. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b, 1988a, 1989a, 1990).

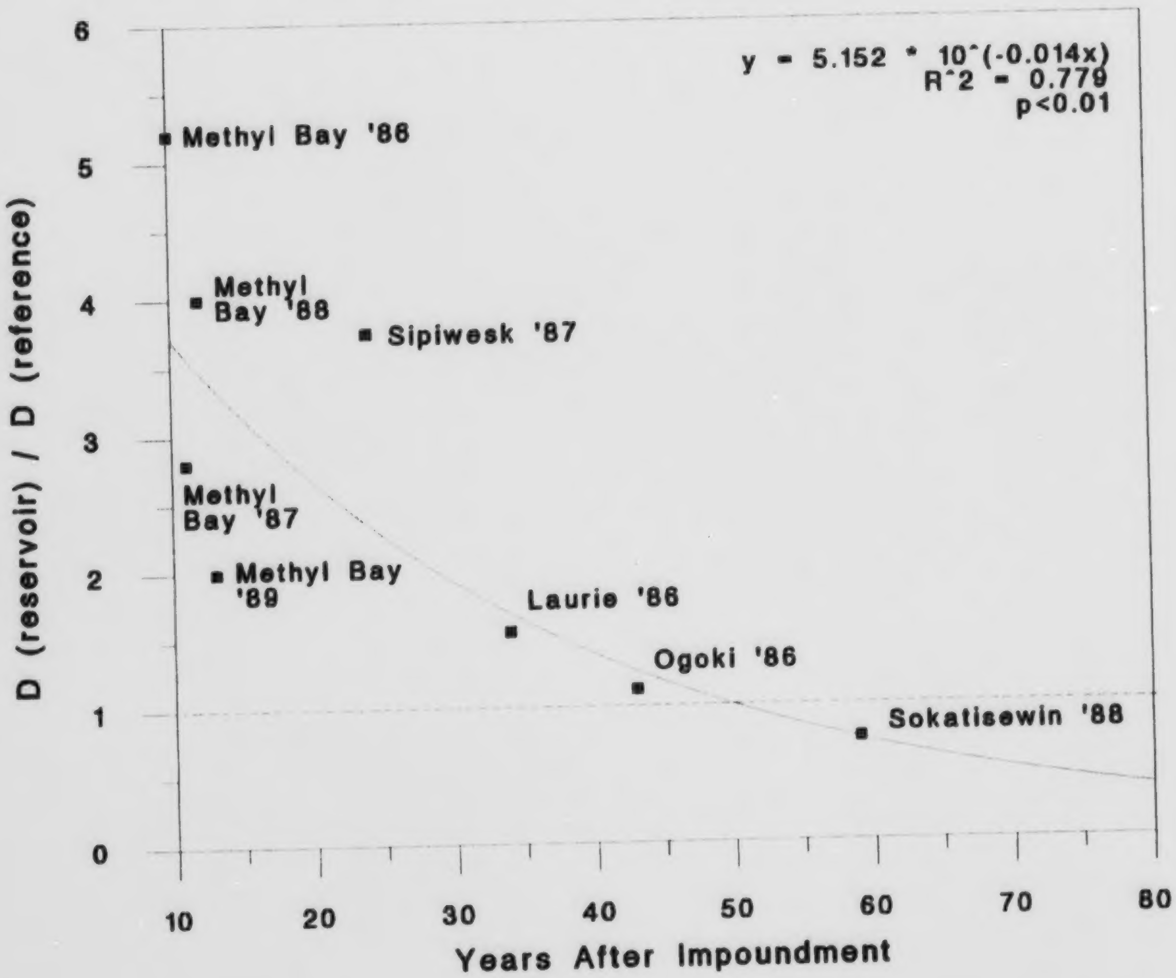

Figure 30. Relationship between the elevation in seasonal mean specific demethylation rate (% added Hg demethylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed between 1986 and 1989. Dashed line represents equality of rates in reservoir and reference lake sediments. Granville Lake was the reference site in all years. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b, 1988a, 1989a, 1990).

return of methylation rates to the reference level is 78 years after impoundment, compared with 53 years for demethylation to return to baseline (Figures 29 and 30).

The methylation balance decline curve in Figures 31 and 32, was derived from the regression lines for methylation and demethylation in Figures 29 and 30. The least squares means of the elevated methylation and demethylation values were calculated at 10 year intervals for the period from 10 to 80 years after impoundment. These values were used to calculate the corresponding M/D values to which the curve in Figures 31 and 32 was fitted. As found for the 1986 data alone, a second order polynomial regression provided the best fit to the predicted M/D values. This model indicates there will be a slowly accelerating rate of decline in net methyl mercury production over time. The half-time (i.e., the period required for a 50% reduction) for the elevated methylation balance in the CRD is estimated to be 59 years, returning to the pre-development level 78 years after impoundment.

Although the basic structure of the models has remained the same over the course of the FEMP studies, there have been two important modifications. First, the dependent variables incorporated in the models have been modified from the measured specific methylation and demethylation rates (Ramsey 1987a and 1988a) to the elevation in methylation, demethylation, and M/D (Ramsey 1989a, 1990). Elevation is calculated as the seasonal mean value in flooded sediments from a given reservoir site divided by the seasonal mean value at the reference site. Granville Lake

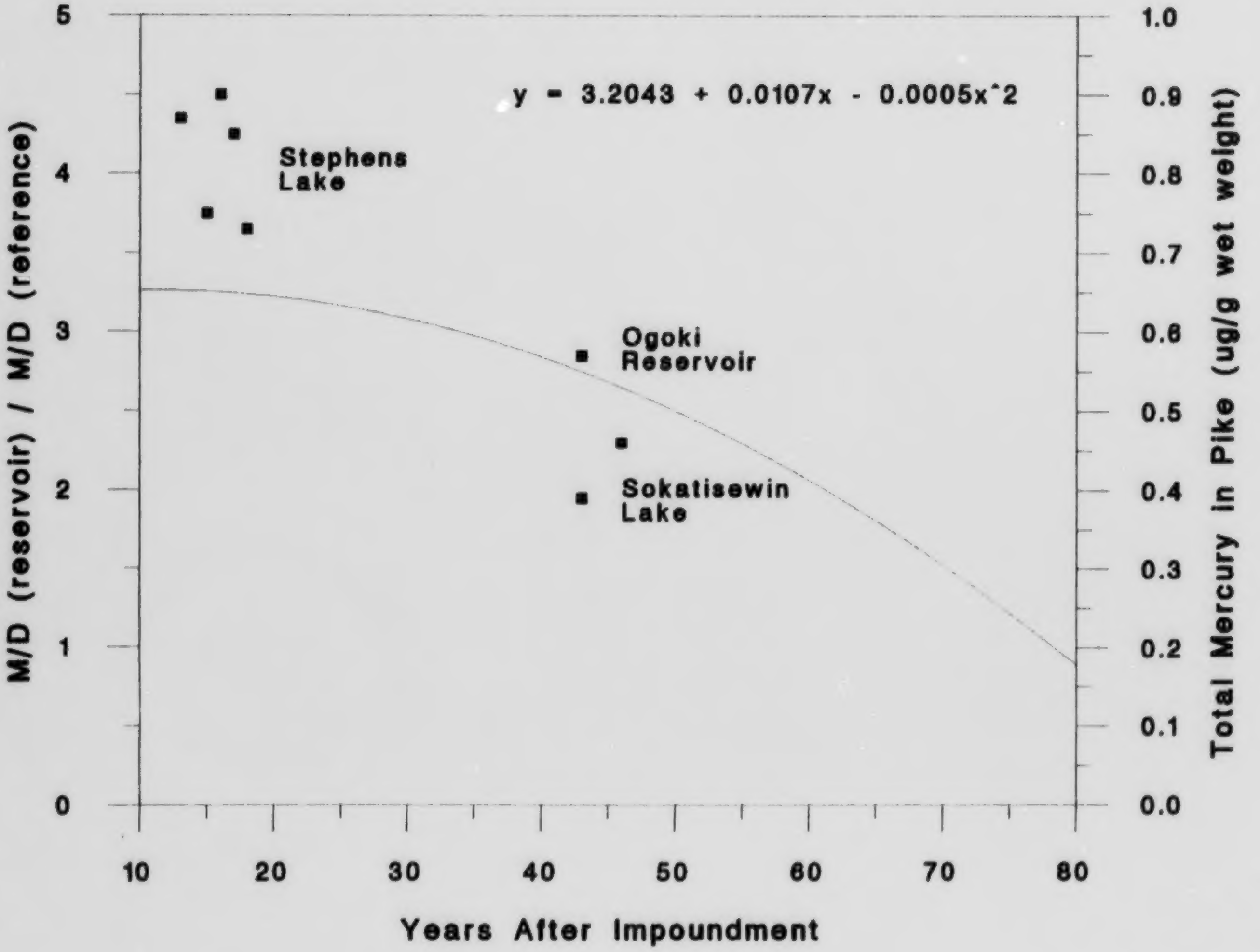

Figure 31. Relationship between elevation in methylation balance (M/D), mean mercury concentration in northern pike, and reservoir age for sites surveyed between 1986 and 1989. The M/D decay curve was calculated from the best-fit lines in Figures 29 and 30. Pike mercury data for Stephens Lake are length-standardized (550 mm) values from Green (1990). Ogoki Reservoir data are length-standardized (550 mm) values from G. Pope, Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario. Sokatisewin Lake data are from commercial fishery samples as reported by Murray (1978).

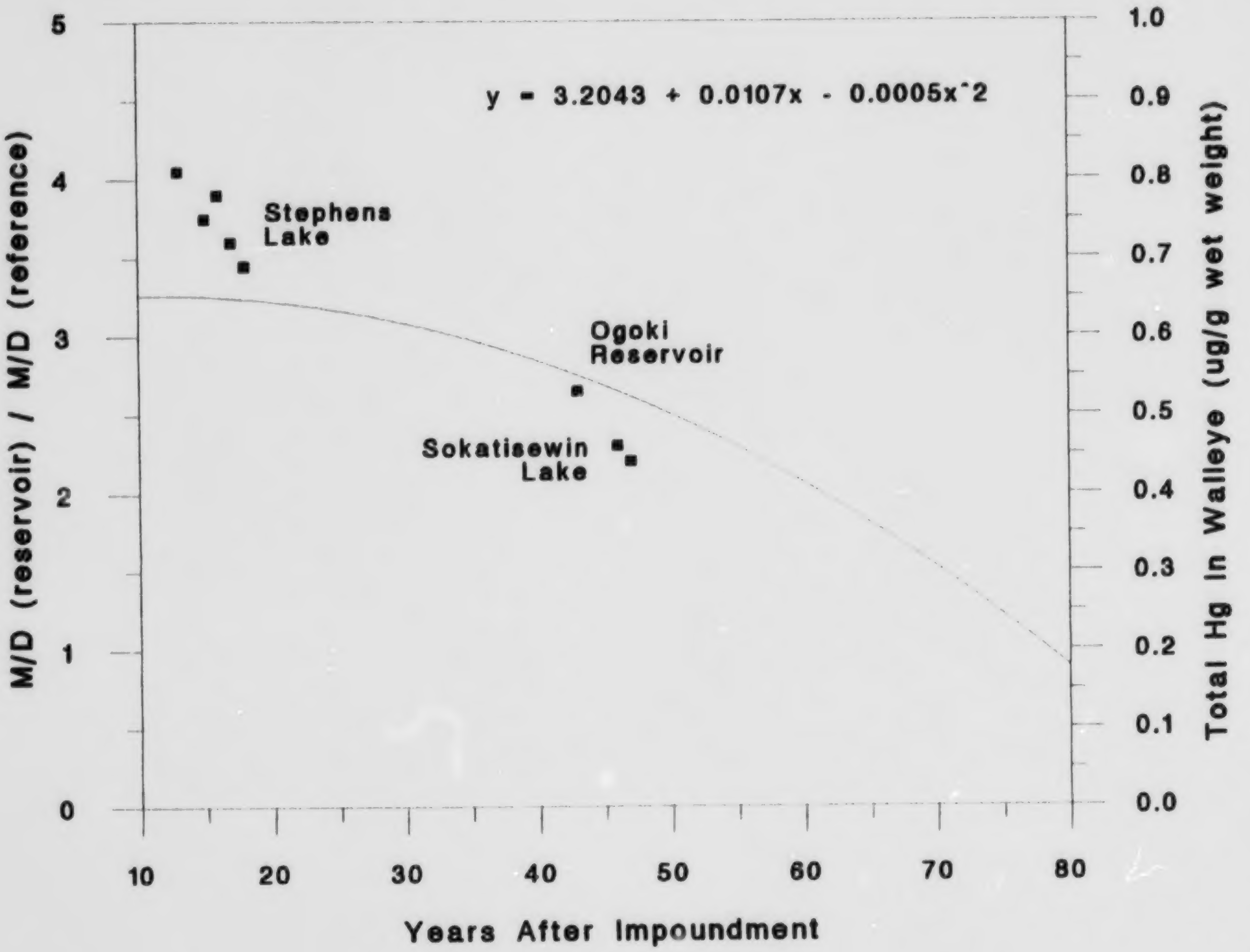

Figure 32. Relationship between elevation in methylation balance (M/D), mean mercury concentration in walleye, and reservoir age for sites surveyed between 1986 and 1989. The M/D decay curve was calculated from the best-fit lines in Figures 29 and 30. Walleye mercury data for Stephens Lake are length-standardized (400 mm) values from Green (1990). Ogoki Reservoir data are length-standardized (400 mm) values from G. Pope, Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario. Sokatisewin Lake data are from commercial fishery [illegible]

served as the reference site for all years. This standardization of the data was necessary to allow comparison of measurements made over several years because the specific methylation and demethylation rates are relative rather than absolute measures of methylating and demethylating activity.

The second and more recent change in the models is related to the nature of the mathematic expression employed to describe the declines in methylation and demethylation rates over time. The models presented in Ramsey (1989a and 1990) were based on regressions of ln-transformed elevation in methylation and demethylation against ln-transformed reservoir age (i.e., a ln-ln plot). The latest models presented in Figures 29 and 30 are the equivalent of regressions of ln-transformed methylation and demethylation against non-transformed reservoir age (i.e., a semi-ln plot). This second change was made because the revised models provided a slightly better fit to the methylation data. The change in the base exponent of the regression equations, from base-10 to base-e, had no effect on the shape or goodness-of-fit of the models.

All the seasonal mean values used in the models represent the time-weighted mean of five measurements, made at approximately 2 week intervals, for the period beginning June 24 and ending September 3. With the exception of the data for Methyl Bay in 1986, which represents a single station, all plotted values represent the average of data from three stations at each site.

The validity of the M/D decline model as a means of predicting the rate of decline of fish mercury levels was evaluated by overlaying pike and walleye mercury data on the predicted M/D decline curve. All fish mercury data were from reservoirs with similar characteristics which included; the same major fish species, the absence of an upstream reservoir within 130 km, similar extent of flooding, comparable regional or pre-development baseline mercury levels, and location within the boreal forest zone. These criteria were selected because they permit the assumption that peak post-impoundment fish mercury levels would have been similar. The distance of 130 km from the nearest upstream reservoir was based on the study of methyl mercury in water along the CRD, in which the export of methyl mercury from Notigi Reservoir was found to be restricted to a distance shorter than the 130 km between the Notigi Dam and Thompson (Section 2.4; Ramsey 1990).

The only Canadian reservoirs which appear to satisfy these requirements are Stephens Lake on the Nelson River (flooded 1970-71, 300% increase in lake area), the Ogoki Reservoir in north-western Ontario (flooded 1943, 266% increase in lake area, Ramsey 1987b), and Sokatisewin Lake on the Churchill River (flooded 1929, approximately 200% increase in lake area).

There is good agreement between the predicted elevation of M/D in a reservoir of given age, and the measured mercury concentration in fish (Figures 31 and 32). The slopes of the declines in M/D and fish mercury are similar and fish mercury levels appear to remain elevated for at least 47 years after impoundment. It has already been shown that mercury levels in Stephens Lake pike and walleye were 3-4 times above pre-development values 18 years after impoundment (Section 4.2.3). There also is strong evidence to suggest that levels in Sokatisewin Lake fish were elevated as late as 1976, 47 years after impoundment. No pre-development data exist for the lake, but data from other lakes on the Churchill River in northern Saskatchewan, upstream of Sokatisewin Lake, indicate fish mercury levels are considerably higher than is characteristic for the system (Table 8). Mercury levels in pike and walleye from upstream lakes ranged between 0.02 and 0.32 $\mu g\ g^{-1}$ and averaged 0.16 and 0.17 $\mu g\ g^{-1}$, respectively, for the period between 1970 and 1976. In the same period, mean walleye and pike mercury levels in Sokatisewin Lake ranged between 0.44 and 0.57 $\mu g\ g^{-1}$ and were 2.8 and 3 times higher than the respective upstream means. The lowest levels in Sokatisewin pike and walleye were, respectively, 60 and 38% above the highest levels measured upstream. Mercury levels in fish from Sokatisewin Lake have not been sampled since 1976 (G. McGregor, Department of Fisheries and Oceans, Inspection Services Branch, Winnipeg, MB, pers. comm.).

There is less certainty regarding the status of fish mercury levels in the Ogoki Reservoir. The river system has not been extensively sampled and there are no pre-development data for the lake. However, the mean methylation balance in Pikitigushi Lake, a nearby reference lake (Figure 5), was within 4% of that in Granville Lake in 1986 (Ramsey 1987b) and the mercury levels in

Table 8. Total mercury concentrations ($\mu g\ g^{-1}$ wet weight) in fish from lakes on the Churchill River in northern Saskatchewan. All data from commercial fishery samples as reported by Murray (1978). The contribution of a given lake-year to the overall mean was weighted by the number of samples in that year.

		Walleye		Northern Pike	
Lake	Year	Total Hg	n	Total Hg	n
Uskik	1971	0.21	5	0.18	2
	1972	0.08	1	0.29	1
Trade	1971	0.32	5	0.18	6
	1972	0.09	1	0.20	2
	1976	0.16	1	0.17	1
Pinehouse	1971	0.14	6	0.21	6
	1972	ND	ND	0.09	1
	1973	0.14	1	ND	ND
Ile-a-la-Crosse	1970	0.12	1	ND	ND
	1971	0.14	5	0.16	11
	1972	0.17	1	ND	ND
	1975	ND	ND	0.24	1
Churchill	1970	0.05	1	ND	ND
	1971	0.10	7	0.14	5
	1972	0.03	1	0.02	1
	1975	ND	ND	0.18	1
Frobisher	1970	0.08	1	ND	ND
	1971	0.18	5	ND	ND
	1975	0.12	1	ND	ND
Weighted Mean		0.16		0.17	
Sokatisewin	1972	ND	ND	0.57	6
	1975	0.46	1	0.46	1
	1976	0.44	2	ND	ND

ND No data.

walleye from the two lakes were the same (Pikitigushi, 0.27 $\mu g\ g^{-1}$; Ontario Ministry of Natural Resources, unpubl. data; Granville, 0.26 $\mu g\ g^{-1}$; Inspection Services, Department of Fisheries and Oceans, Winnipeg, MB.). This indicates the relationship between methylation balance and fish mercury in the Ogoki Reservoir area is the same as in the CRD, suggesting that M/D and fish mercury in the reservoir are elevated above natural baseline.

The estimates of recovery time presented above are considerably shorter than indicated previously, particularly for methylation and the methylation balance. Using the same data, I calculated a recovery time of approximately 135 years (Ramsey 1990), compared with the 78 year duration calculated above. This difference was due to the changes in the regression expression described above. The comparison of the decline in M/D with mercury levels in fish (Figures 31 and 32) suggests the revised model provides a better description of the time course.

4.4 Factors Affecting Duration

In the course of studies on mercury in the NFA area, a number of factors have emerged as having a significant modifying effect on the duration of mercury problems in northern reservoirs. These include the nature of the flooded organic matter with respect to its susceptibility to decomposition, the presence of a continuing source of fresh organic matter, and the erosion of shoreline clays.

The importance of the type of organic matter flooded is best illustrated by the time course of the elevated fish mercury levels in Cedar Lake following impoundment by the Grand Rapids Generating Station in 1965. Within 12 to 14 years after flooding, mercury concentrations in pike and walleye had declined to levels commonly found in northern Manitoba lakes remote from hydro development. The major difference between Cedar Lake and flooded lakes on the Rat-Burntwood and Nelson river systems was in the type of vegetation flooded. The area of Cedar Lake approximately doubled and about 80% of the flooded land was located in the area of the Saskatchewan River Delta (Green and Derksen 1980). Marsh macrophytes and sedges dominated the delta prior to flooding. These materials decompose more quickly than the boreal forest vegetation flooded in the other reservoirs, and contribute much (est. 90%) less organic matter per unit area of flooded land (Ramsey and Ramlal 1987b). These features of the Cedar Lake impoundment contributed to a much faster recovery than has occurred in any of the reservoirs in the NFA area.

Another factor to be considered is the way in which reservoir levels are regulated. In Sipiwesk Lake, levels are generally low in summer and high in winter. This is the reverse of the situation in most hydroelectric reservoirs, where water is stored during the summer and released in winter to meet increased power demands. Sipiwesk Lake was regulated in the usual manner until 1976, when the Jenpeg dam began operating, producing the reversed level regime. One of the results of this regime is exposure of the drawdown zone in summer when it can become recolonized by grasses and other fast-growing vegetation. These materials are flushed into the lake during periods of high water, becoming available to the methylating bacteria. A continuous source of fresh organic matter such as this should maintain methyl mercury production at a high level longer than would be expected in a reservoir with the usual level regime. This appears to be the case in Sipiwesk Lake, where the specific methylation rate in the flooded zone was at the high end of the expected range in 1987 (Ramsey 1989a).

The long-term impact of eroding glacial clays on the duration of the mercury problem in the CRD remains to be established. It was suggested in Section 4.2.5 that the decreases in fish mercury levels which have occurred in Rat and Stephens lakes coincided with increases in shoreline erosion. It was speculated the eroding clays were covering some of the flooded organic matter, effectively reducing the area of reservoir sediments supporting elevated methylation and resulting in lower mercury levels in fish. There is insufficient information to determine if this process will cause a continuing reduction in fish mercury levels, accelerating recovery of the mercury problem.

4.5 Duration of Mercury Problems in other Regions

The projected duration of the mercury problem in SIL and along the CRD, 78 years following flooding, is much longer than the predicted duration for the problem in the Complexe La Grande in northern Quebec. Verdon (1990) estimated mercury levels in northern pike from the Complexe La Grande will return to the baseline level 20-30 years after impoundment. This estimate is based on a comparison of mercury concentrations in pike from reservoirs in

Quebec and Labrador ranging in age from 3 to 67 years (Table 9). There is clear evidence of elevated levels continuing in reservoirs up to 16 years of age, the oldest reservoir in the data set for which there are pre-development data. The mercury concentration in 700 mm pike from Smallwood Reservoir in Labrador, at 1.11 $\mu g\ g^{-1}$ in 1988, was still 2.6 times higher than the pre-development level after 16 years of impoundment and had declined just 0.16 $\mu g\ g^{-1}$ in the preceding 10 years.

The estimated recovery time for the mercury problem in the Complexe La Grande relies heavily on data from older reservoirs for which no pre-development data exist. Mercury levels in pike from reference lakes adjacent to the older reservoirs were instead used as an indication of pre-development values. Only 1 reference lake was sampled for each of the older reservoirs, and no rationale for the selection of these reference lakes has been presented. Curiously, mercury levels in fish from the reference lakes were extraordinarily high, ranging between 1.06 and 1.38 $\mu g\ g^{-1}$ for 700 mm northern pike, compared with 0.43 to 0.63 $\mu g\ g^{-1}$ in the reservoirs for which pre-development data were available (Table 9). Without knowledge of the rationale used in selection of the reference lakes, and considering the extremely high mercury levels reported for the lakes, it is difficult to accept them as valid indicators of pre-development fish mercury levels. There is considerable natural variation in fish mercury levels among lakes such that, in the absence of pre-development data for a given site, very different conclusions can be drawn depending on the reference

Table 9. Total mercury concentrations ($\mu g\ g^{-1}$) in northern pike (700 mm) in reservoirs of varying ages in Quebec and Labrador. Natural Hg levels for LG-2 and Smallwood reservoirs are pre-development values. For other sites, natural levels are concentrations in a single nearby reference lake. Augmentation factor calculated as Total Hg/Natural Hg. Data from Verdon (1990).

Reservoir	Age (Years)	Total Hg ($\mu g\ g^{-1}$)	Natural Hg Levels ($\mu g\ g^{-1}$)	Augmentation Factor
LG-2 (1982)	3	1.32	0.63	2.1
LG-2 (1984)	5	2.66	0.63	4.2
LG-2 (1986)	7	2.30	0.63	3.7
LG-2 (1988)	9	3.14	0.63	5.0
Smallwood	6	1.27	0.43	3.0
Smallwood	16	1.11	0.43	2.6
Manic-5 (1985)[1]	20	1.94	1.23	1.6
Dozois	37	1.43	1.38	1.0
Decelles	44	0.99	1.38	0.7
Baskatong	59	1.37	1.06	1.3
Des Quinze	62	1.13	1.38	0.8
Gouin	67	1.19	1.22	1.0

[1] Flooding of Manic-5 occurred from 1964 to 1970.

lake which is selected. For example, in 11 non-flooded reference lakes within 60 km of SIL, mercury levels in walleye range from 0.23 $\mu g\ g^{-1}$ in Gauer Lake, which is similar to the pre-impoundment value for SIL, to 0.86 $\mu g\ g^{-1}$ in Melvin Lake (Rannie and Punter 1987), which is higher than the peak post-impoundment walleye mercury value for SIL (Figure 24). A similar degree of inter-lake variation should be expected in Quebec as well, making any conclusions based on a single reference lake rather tenuous. Thus, it appears there is little justification for the claim that fish mercury levels in the Complexe La Grande will return to pre-development values within 20-30 years of impoundment.

The lack of pre-development fish mercury data for reservoirs in excess of 20 years of age makes it necessary to infer pre-development concentrations from nearby reference lakes. The substantial natural variation in fish mercury levels among lakes makes it difficult to arbitrarily select a single lake as representative of baseline levels. Considerably more data on baseline mercury levels in lakes around the older (>20 years of age) Quebec reservoirs are required before it can be concluded the reservoirs have recovered.

There is little reason to expect that the fish mercury problem in the Complexe La Grande will be of substantially shorter duration than the problem on the CRD. The two developments are situated in very similar biophysical settings and the same factors appear to be regulating the microbial methylation balance (Ramsey 1989b). The only major biophysical difference is the general absence of glacial clays in the La Grande area, resulting in little continuing shoreline erosion. This difference is not expected to shorten the duration in La Grande. No active shoreline erosion is occurring in Methyl Bay on SIL and neither of the two oldest reservoirs sampled in the FEMP studies, Ogoki and Sokatisewin Lake, is situated in an area of glacial clay deposits. Nevertheless, fish mercury levels in the older reservoirs fit the M/D decline curve (Figures 31 and 32).

The predicted duration of the mercury problem in the CRD also is considerably longer than reported for reservoirs in Finland. Verta et al. (1986) found a half-life of 15-20 years for elevated pike mercury levels in 20 reservoirs, ranging from 3 to 20 years in age. In most of the reservoirs, pike mercury was near the regional average of 0.56 $\mu g\ g^{-1}$ within about 15 years of impoundment. However, levels in some reservoirs remained high (>1.0 $\mu g\ g^{-1}$) 16-18 years after impoundment. The major difference between the Finnish reservoirs and those in northern Manitoba, and the likely explanation for the shorter time-course in Finland, is in the level management regimes. The Finnish reservoirs are typically shallow, with mean depths between 1.4 and 7.0 m, and are heavily regulated. The ratio of drawdown to mean depth ranged between 0.4 and 2.29 and averaged 1.16. Most are run off in the winter until almost empty and are refilled by the spring floods. The flooded materials are therefore subjected to substantial physical disturbance by ice and wind which would accelerate the processing and removal of organic matter from the reservoir. This physical processing does not occur to the same extent in northern Manitoba reservoirs due to the much more limited range of level fluctuation relative to mean depth. The maximum annual range of levels on SIL is approximately 1 m compared to a mean depth of 9.8 m (Newbury et al. 1984) for a drawdown ratio of 0.1.

5.0 Toxicity of Elevated Mercury Levels to Biota

Some concern has been expressed by users of fish and aquatic furbearer resources along the CRD about the possible effects of the elevated mercury levels on animal physiology and behaviour (L. Linklater, Nelson House, MB, pers. comm.). The available information indicates it is unlikely that physiological effects on fish, mink, or otter are now occurring or have occurred in the past, even at the highest concentrations measured on the CRD. Mercury concentrations in other mammals and birds from the NFA area are unknown, so it is not possible to assess the potential for physiological effects in these groups.

Laboratory toxicity studies on brook trout (*Salvelinus fontinalis*) indicate that whole-body mercury concentrations would have to exceed 3 to 5 $\mu g\ g^{-1}$ before physiological effects should be expected. McKim et al. (1976) found levels up to 2.7 $\mu g\ g^{-1}$ had no effect on growth or survival of juvenile or adult brook trout nor on reproduction, while levels in the range of 5 to 7 $\mu g\ g^{-1}$ eventually produced symptoms of toxicity and death. Olson et al. (1975) found concentrations up to 10.9 $\mu g\ g^{-1}$ in fathead minnows (*Pimephales promelas* Rafinesque) had no effect on the survival, growth, behaviour, or general appearance of the fish. Lockhart et al. (1972) found differences in the biochemical profiles of northern pike (*Esox lucius*) from Clay Lake, Ontario, with mercury levels in the range of 6.3 to 16.0 $\mu g\ g^{-1}$, when compared with pike from Heming Lake, Manitoba, with mercury levels of about 0.3 $\mu g\ g^{-1}$. These differences indicated the Clay Lake fish were stressed, but this could not be attributed directly to the high mercury levels because the lakes differed in other ecological factors.

Mercury concentrations in fish along the CRD presently are below the levels which have been shown to produce physiological effects. Mean mercury concentrations in pike and walleye from Rat and Threepoint Lake are in the range of 1.0 to 1.2 $\mu g\ g^{-1}$ and the maximum level found in an individual fish in 1989 was 3.31 $\mu g\ g^{-1}$ in a pike from Rat Lake (Green 1990). In the past, mercury levels in some of the largest fish may have approached the upper limit of the no effects range. In 1983, the highest mercury level measured in an individual walleye from Rat Lake was 4.1 $\mu g\ g^{-1}$ (Green 1990). The highest mercury concentration ever found in an individual fish from the CRD was 4.73 $\mu g\ g^{-1}$ in a pike from Karsakuwigamak Lake captured in 1984 (Green 1986). Considering that laboratory studies required higher fish mercury concentrations than the highest recorded in any single fish from the CRD to produce any measurable physiological effects, and that no clear evidence of physiological effects has been found in the field at even higher concentrations than in laboratory studies, there appears to be no justification for the examination of mercury-related physiological effects in the CRD in any more detail.

Kucera (1987) reported that mean mercury concentrations in tissues of mink and otter also were well within the non-toxic ranges for these species. The highest concentrations measured in individual animals approached the upper limit of the non-toxic range, as in fish.

6.0 Mitigation Measures

The following actions have been identified in the literature as possible approaches to the mitigation of aquatic mercury problems or rehabilitation of closed fisheries:

- selenium additions
- clay resuspension (increased turbidity)
- increase fish growth
    - intensive fishing
    - nutrient additions
- lake liming
- inhibition of methylation/stimulation of demethylation
- make use of species or size-classes of fish with lower mercury concentrations

6.1 Selenium Additions

The addition of selenium has been suggested as a potentially successful means of reducing fish mercury levels both in Canada and Scandinavia. Selenium has been shown to reduce the toxicity of several heavy metals, including mercury (Frost and Lish 1975, Vokal-Borek 1979), and can reduce the severity of inorganic and methyl mercury poisoning. Turner and Rudd (1983) found selenium additions also can reduce mercury bioaccumulation in aquatic food webs. These decreases can be substantial. Paulsson and Lundbergh (1990) reported a 50% decrease in pike mercury levels, and a 90% decrease in perch and roach mercury after less than a year of treatment with selenium at 3-5 $\mu g\ L^{-1}$.

A serious difficulty with the use of selenium is that the levels in water which are required to produce a significant reduction of mercury levels in fish (1-5 $\mu g\ L^{-1}$: Turner and Rudd 1983, Paulsson and Lundbergh 1990) come uncomfortably close to the Canadian limit for drinking water (10 $\mu g\ L^{-1}$; Health and Welfare Canada 1989). It is unlikely that selenium additions will be a viable means of ameliorating the mercury problem in the CRD for the foreseeable future unless the effective level can be substantially reduced or it can be conclusively demonstrated that the levels which have been reported to be effective are non-toxic to humans or aquatic life.

6.2 Increased Turbidity

Sediment resuspension is the only remedial measure which has been tested for application on the CRD. This method has been suggested as a possible mitigative action for the mercury problem in the English-Wabigoon river system in northwestern Ontario on the basis of a limnocorral experiment which indicated the method could effectively reduce mercury levels in fish (Rudd and Turner 1983). A similar study in SIL found otherwise. Increased suspended sediment concentrations did not significantly reduce mercury concentrations in fish and caused slight increases in some cases (Hecky et al. 1987a).

The conflicting results of these studies could be a reflection of the different causes of the problems in the two systems; increased inorganic mercury loading in the English-Wabigoon compared with increased organic matter loading in the CRD. The suspended sediments were thought to reduce mercury accumulation in fish by binding inorganic mercury, making it unavailable for methylation (Rudd and Turner 1983), and possibly binding methyl mercury as well, making it unavailable for uptake by fish. With the problem in the CRD apparently not due to increased inorganic mercury availability, control of this variable should not be expected to have a significant effect on methylation.

The discrepancy may also be a result of different experimental approaches. In the English-Wabigoon study, ^{203}Hg was added as a radiotracer and only ^{203}Hg was measured in the fish. An increase or decrease in ^{203}Hg concentration in tissue was assumed to be indicative of a change in total mercury bioaccumulation. The radiotracer also was added in the SIL experiments, but bioaccumulation rates of both total mercury and radio-labelled mercury were measured. The SIL study indicated that, in short term experiments, radiomercury uptake or loss was not necessarily indicative of total mercury uptake or loss (Hecky et al. 1987a). Thus there is serious question regarding the efficacy of sediment resuspension as a mitigating measure in any system, and it certainly does not seem to be applicable in the CRD. Indeed, the mercury problem developed at most locations in spite of increased suspended sediment concentrations. Although increased turbidity was not effective in reducing mercury bioaccumulation by fish, the covering of organic matter with inorganic sediments should still be effective in reducing microbial methylation (Hecky et al. 1987b).

6.3 Increase Fish Growth

The application of measures to increase fish growth as a means of reducing the mercury concentration in tissues has been suggested as a way of mitigating mercury problems by a number of authors (e.g., Beijer and Jernelov 1979; Jones et al. 1985). It is thought that if fish growth can be increased without causing a proportionate increase in methyl mercury availability, then the mercury concentration in fish flesh will be reduced. If mercury burden is related to methyl mercury availability, then the burden will be diluted in faster growing fish leading to lower mercury concentrations. This biodilution concept has been criticized, but the majority of evidence in the literature suggests the concept has merit at least under some circumstances. Two methods of increasing fish growth, their impact on mercury levels in fish and potential utility in the CRD are discussed below.

6.3.1 Intensive Fishing

Intensive fishing has been suggested as a means of reducing fish mercury levels by increasing fish growth rates, with some positive effects demonstrated in Finland. Iivonen and Verta (1990) intensively fished Hakojärvi, a small (0.17 km^2) forest lake in southern Finland in 1984 and 1985. More than half the fish stock, 29.5 kg ha^{-1}, mainly roach (*Rutilus rutilus* L.), was removed from the lake. The mean standardized (1 kg) mercury concentration in northern pike (*Esox lucius*) declined from 0.97 $\mu g\ g^{-1}$ before harvesting to 0.45 $\mu g\ g^{-1}$ in 1987. Similarly, the mean standardized (0.5 kg) level in burbot (*Lota lota*) decreased from 0.56 to 0.19 $\mu g\ g^{-1}$ and the mean level in roach (50 g) dropped from 0.36 to 0.25 $\mu g\ g^{-1}$ during the same period. The mercury concentration in perch (*Perca fluviatilis*) did not change between harvest and 1987. Since 1987, mercury levels in pike, burbot, and perch have increased, although mercury in pike and burbot had not returned to pre-harvest levels by 1989. Growth rates increased in all species, with the largest increase in pike and the smallest in burbot. Growth rates of perch and roach increased initially but returned to pre-harvest rates within three years. The reduced fish mercury concentrations were attributed to the increased growth rates (pike, roach) and altered diet (burbot).

Recent research has shown that a large amount of the methyl mercury in a lake, about 30%, is tied up in the fish (J. Wiener, Wisconsin Department of Natural Resources, La Crosse, Wisconsin, pers. comm.). This suggests that the removal of a significant portion of the fish community, through intensive fishing, might also reduce the size of the methyl mercury pool, further contributing to the reduction of mercury levels in the remaining fish.

Continued research on the mechanism by which intensive fishing reduces mercury levels appears to be warranted.

Although intensive fishing would appear to be an effective means of reducing mercury concentrations in fish, at least for periods of 3 to 5 years, its usefulness as a mitigating measure in northern Manitoba reservoirs remains to be established. An important limitation is the impact of the harvest on fish populations and their ability to support further harvest afterward.

Intensive fishing also is subject to some practical limitations. The lakes must be small enough for it to be feasible to substantially reduce fish standing stocks, and the potential for immigration of contaminated fish from other parts of the system must be limited. Any trophy sport-fishery on the lake also would be lost. These restrictions would exclude most of the lakes comprising Notigi Reservoir, with the possible exception of Central Mynarski Lake. Lakes below the Notigi Dam are better candidates for manipulation, provided fish from Notigi Reservoir do not contribute significantly to downstream stocks. This remains to be established.

Another potential use of intensive fishing is in rehabilitation of the "inland" lakes with naturally high mercury levels which are common in the NFA area. These lakes are generally smaller and better isolated than those on the CRD, making them more suitable for manipulation. Impacts on sport fisheries also are more easily avoided. Because of these characteristics, the inland high mercury lakes might be a better site to test the effectiveness of intensive fishing than lakes on the CRD.

6.3.2 Nutrient Additions

Another means of increasing fish growth, and potentially lowering muscle mercury concentrations, is the stimulation of algal productivity through inorganic nutrient additions. The best evidence that this approach might work is reported by D'Itri at al. (1971). They compared mercury concentrations in stocked rainbow trout from an **oligotrophic** and a **eutrophic** lake in northern Michigan. The mercury concentration in fish from the eutrophic lake was less than half the level in fish from the oligotrophic lake. Fish of a given age from the eutrophic lake weighed twice as much and were 27% longer than fish from the oligotrophic lake, suggesting the higher growth rate was responsible.

An experimental investigation of the effect of nutrient additions on fish mercury levels in SIL was inconclusive. Hecky et al. (1987a) increased **primary productivity** in a limnocorral by a factor of 2, with no change in mercury bioaccumulation by fish compared to a reference enclosure. Fish growth was not affected by the increased productivity so this experiment neither confirms nor refutes the hypothesis of growth dilution.

Regardless of its potential impact on fish mercury levels, nutrient additions would not be useful in mitigating the mercury problem on the CRD. Primary productivity in these turbid lakes is generally light limited (Hecky and Guildford 1984; Ramsey et al. 1991), so nutrient additions will not have a significant impact on primary productivity or fish growth.

6.4 Liming

The liming of lakes through the addition of limestone has been proposed as a solution to mercury problems related to lake acidification. Although liming has been shown to cause some reduction of fish mercury levels in acidified lakes in Sweden, that decrease is small, on the order of 0.3 $\mu g\ g^{-1}$ for a 1 kg pike with a pre-liming mercury concentration of 1.5 $\mu g\ g^{-1}$ (Anderson and Borg 1990). There also is no reason to believe liming would be effective in northern Manitoba reservoirs because the mercury problems on the Rat-Burntwood and Nelson river systems were not caused by a pH reduction.

6.5 Inhibition of Methylation/Stimulation of Demethylation

One of the areas of research identified in the final report of the Canada-Manitoba Mercury Study Agreement was the investigation of methods to selectively stimulate the demethylating bacteria or to inhibit the methylators through the addition of a suitable substrate. This approach requires knowledge of the causative microorganisms and their metabolic pathways. A basic requirement is that the methylators and demethylators belong to different metabolic groups. The FEMP mercury studies included investigations to identify the major metabolic groups of bacteria which were responsible for methylation and demethylation in order to determine if the essential requirement of this approach was satisfied.

A pilot study in 1987 examined the role of methanogens in methylation and demethylation on one occasion during the routine summer sampling (Ramsey 1988a). The methanogens were found to account for the majority of methylation. Methanogen inhibition produced a 76% decrease in the methylation rate in flooded sediments. Demethylation rates increased with methanogen inhibition. This suggested the methylators and demethylators belonged to different metabolic groups, meeting the essential requirement for independent manipulation.

The results of the pilot study were sufficiently promising to warrant further investigation in the summer of 1988, when both methanogens and sulfate reducers were considered. The results of this more extensive investigation were disappointing. When the entire methylating season was considered, it was found that methanogens and sulfate reducers were the only important contributors to total methylation and demethylation in Methyl Bay flooded sediments and that the two groups were equally important in both processes (Table 1). Although a group may dominate one of the processes, but not the other, at a particular time and place, both groups make equal contributions over the season on average. Consequently, it may not be a simple matter to inhibit methylation or stimulate demethylation without having the same effect on the competing process.

6.6 Use Fish Species or Size-Classes with Lower Mercury Levels

It has been known for some time that the different fish species accumulate mercury to varying levels due to differences in feeding habits. Knowledge of these differences has been used to develop public information programs encouraging the domestic consumption of species with lower mercury concentrations. Unfortunately, the species with the lowest mercury levels, usually whitefish and suckers, also have the lowest commercial value. The knowledge that mercury level is dependent upon fish size might be equally useful in the rehabilitation of commercial fisheries based on the higher value predatory species. Small-mesh nets could be used to catch the smaller individuals in the populations of predatory species in lakes where mercury levels in these smaller size classes are acceptable for commercial marketing. Use of a "slot-fishery" was suggested by Jones et al (1985) and came up in discussion with Wayne Wysocki (Symbion Consultants, Winnipeg, MB, pers. comm.) as well. These smaller individuals also are more marketable than larger fish, further contributing to the viability of a commercial fishery. The resumption of commercial fishing in closed lakes might also have some effect in reducing fish mercury levels, as outlined in Section 6.3.1.

Although such a practise normally would not be acceptable to fisheries managers, potentially having a negative impact on future spawning stocks, it might be worthy of consideration in situations where there is very little to lose. One such lake is Wapisu, where standardized mercury levels in walleye and pike of 1.58 and 1.69 μg g^{-1}, respectively, were reported in 1985 (Green 1986). Based on the predicted rate of decline in M/D (Section 4.3; Figures 30 and 31), more than 60 years will be required for the mean mercury levels in these species to drop below the Canadian marketing limit. A "slot" commercial fishery may permit the immediate commercial utilization of this lake and present an alternative to the current practise of the subsidized fishing of cutter lakes. This should be attractive both to the fishermen and the subsidizing agency. Should it prove unviable as a result of stock depletion, then there would still be plenty of time for the stocks to recover as mercury levels in fish naturally decline.

7.0 Prevention of Mercury Problems in New Reservoirs

The current understanding of the cause of the reservoir mercury problem suggests a number of actions which might minimize or prevent the development of a mercury problem in a new reservoir. The dependence of elevated methylation rates on flooded organic matter indicates that some means of removing organic soils and vegetation before flooding, or at least reducing their organic content, should be effective. At this time, the only approach which can be reccmmended is the complete removal of organic matter from a basin before flooding (Hecky et al. 1987b). This measure obviously carries a significant cost and may not be practical for all but the smallest reservoirs. The partial clearing of a reservoir basin might be more feasible, but there is no means of predicting the reduction in peak fish mercury levels which can be expected for a given proportion of clearing.

Another approach which should be examined is the selective clearing of organic materials. Knowledge of which materials are most stimulatory of methylation, or the longest contributors to elevated methylation, would permit the concentration of basin preparation efforts in areas which would provide the greatest benefit. This measure requires information on the ability of specific types of organic matter to stimulate net methylation. Such information is not yet available but could be obtained from limnocorral experiments or field studies comparing the microbial methylation balance among sites with different vegetation cover.

Controlled burning of a reservoir basin prior to flooding might also be used to reduce organic loading, and therefore methyl mercury production. However, combustion may have some undesirable side-effects which remain to be identified.

The covering of bogs and other organic soils with inorganic soils might be used in specific circumstances where vehicle access is possible and there is a ready supply of material. An important technical consideration in this method is the means by which the inorganic cover can be maintained during reservoir filling. Some erosion should be expected as the cover is exposed to wave energy, and it remains to be determined how thick the cover must be to ensure sufficient cover material remains to isolate the underlying organics. This method would probably be inappropriate for use in the drawdown zone or at shallow depths within the wave zone.

It should be emphasized that none of these approaches has been tested in a new reservoir and, therefore, there is no information on which to base a cost-benefit analysis. Given the high profile of environmental mercury problems, their predicted longevity, and the lack of proven mitigation measures, hydroelectric developers should consider researching these prevention options further.

8.0 Prediction of Fish Mercury Levels in New Reservoirs

The prominence of the reservoir mercury problem has stimulated several attempts to develop a means of predicting fish mercury levels in new reservoirs, based on their physical and chemical characteristics, as an aid in assessment of their impact. The extensive fish mercury database and diverse physical features of the impoundments on the Churchill River diversion and Nelson River have made this area the focus of many modelling exercises.

Jones et al. (1985) examined the relationship between mercury concentration in fish and 28 physical and chemical characteristics of 17 reservoirs in Manitoba, Ontario, Quebec, and the Maritimes. They concluded that there was no means of predicting, with any measure of confidence, the occurrence or severity of mercury problems in new reservoirs based on existing knowledge. A number of variables were found to account for significant components of the variance in fish mercury, but too little of the total variance in fish mercury could be accounted for.

The study of Jones et al. (1985) failed to control a number of sources of variation which may have obscured any underlying relationships. The influence of upstream reservoirs was ignored and the temporal trend of fish mercury levels over time also was not controlled. Data for some reservoirs represented near peak values, while samples from other reservoirs were taken near the time of recovery (e.g., Cedar Lake in Manitoba).

Derksen and Green (1987) examined the relationship between mercury levels in fish and the extent of flooding (i.e., the % increase in lake area due to flooding) in lakes on the Churchill River diversion and Nelson River. They failed to find a significant, quantifiable, relationship between the variables which could be used for prediction. Their analysis indicated a positive correlation between the ranks of mercury in fish and extent of flooding, which suggested a cause-effect relationship, but this correlation was of no predictive use. The lack of a direct relationship between extent of flooding and mercury concentration was attributed to sampling variability and the number and complexity of factors which might influence such a relationship.

The efforts of Jones et al. (1985) and Derksen and Green (1987) concentrated on examination of the relationship between mercury levels in fish and within-lake physical and chemical characteristics. Subsequent studies have found the age of the reservoir at the time of sampling, and the influence of upstream reservoirs also must be considered.

Ramsey (1987b) found a significant correlation between mercury concentration in walleye and extent of flooding for lakes on the CRD and Nelson River by incorporating a qualitative measure of the influence of upstream reservoirs. Lakes were divided into 2 groups, those with, and those without a reservoir immediately upstream, and the groups were considered separately (Figure 33). The same approach was unsuccessful for northern pike, probably because the model considered upstream effects as a constant rather than as a

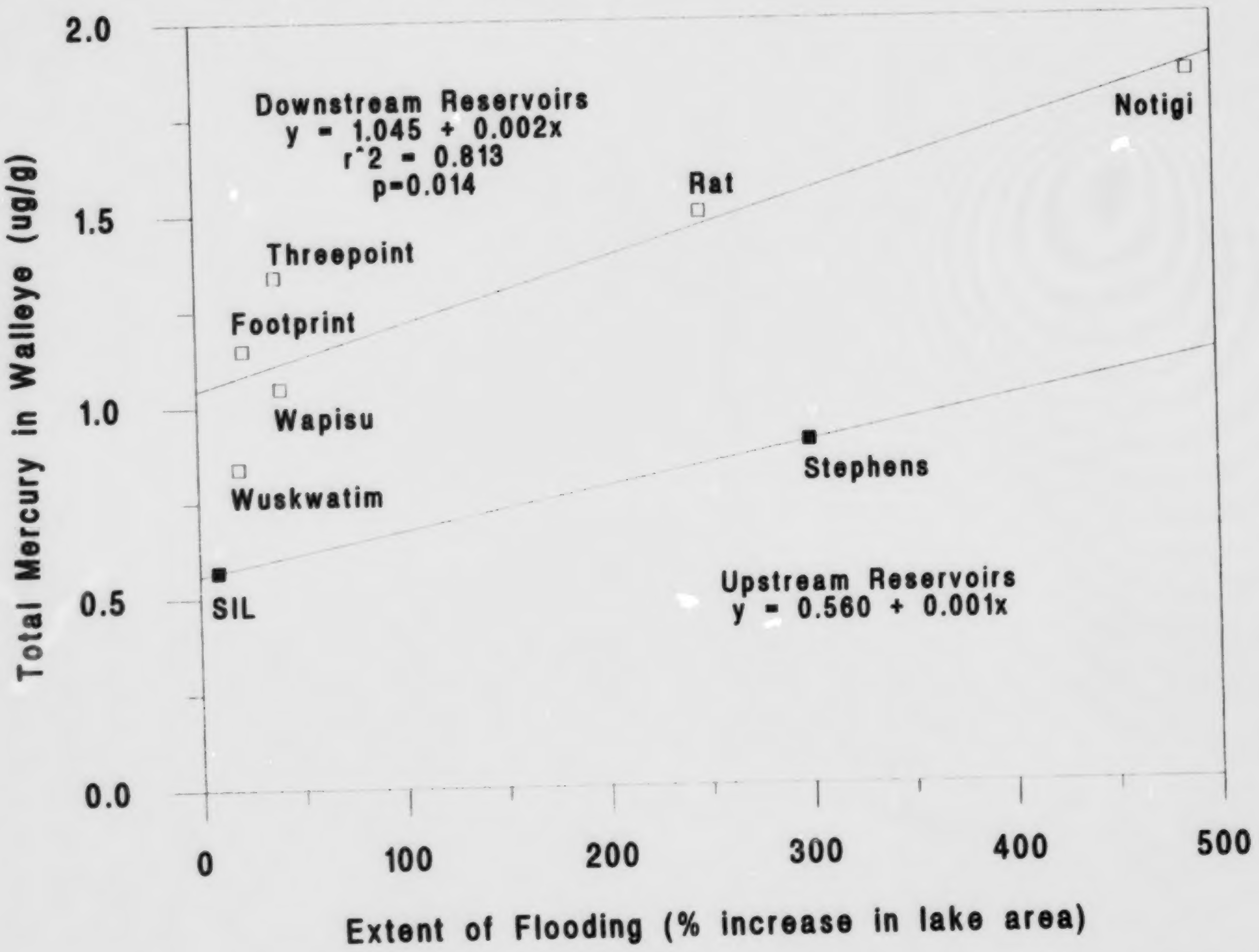

Figure 33. Relationship between extent of flooding (% increase in lake area due to flooding) and mercury concentration in walleye in lakes on the Churchill River diversion and Nelson River, Manitoba. Fish mercury data are means of all lake survey data collected between 1980 and 1984 as reported in Bodaly et al. (1984) and by Department of Fisheries and Oceans, Inspection Services, Winnipeg, MB. Extent of flooding values for all lakes from G. McCullough (Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.). Figure modified from Ramsey (1987b) to reflect the latest (3 February 1992) extent of flooding estimates.

variable.

The importance of upstream reservoirs has since been incorporated in a more general model by Johnston et al. (1991). Once again working with data from the Churchill River diversion and the Nelson River, they found significant positive correlations between the peak post-impoundment, standardized, mercury burden and the rise in water level (RISE), percent flooding (PF), upstream percent flooding (UPF), flooded area to volume ratio (AVR), and the upstream flooded area to volume ratio (UAVR) for whitefish, walleye, and northern pike, the three species examined. Variables representing in-lake changes (RISE, PF, and AVR) accounted for 34-59% of the variation in mercury burden, while upstream changes accounted for 66-76% of the variation. Combining in-lake and upstream effects in a two-variable model accounted for 69-84% of the variation in mercury burden, significantly improving the models for whitefish and walleye but not for northern pike. Models using flooded area to volume ratio variables generally accounted for more of the variation in walleye and pike mercury burden than did models based on percent flooding variables.

The models performed well for flooded lakes on the Churchill River diversion but not in other reservoir systems. The model overestimated peak mercury burden in fish from Cookson Reservoir, Saskatchewan. This was attributed to differences in soils, vegetation, and water chemistry and the potential effect of these variables on microbial methylation. The models also performed poorly for reservoirs in Labrador and northern Quebec. This was considered to be a result of the dominant influence of in-lake effects in these projects. Additional investigation is required to determine the circumstances under which in-lake effects dominate over upstream effects, and vice-versa.

Despite the success of the models in accounting for variation in fish mercury levels among reservoirs on the CRD, it should not be assumed that the models are applicable to all northern Manitoba reservoirs. The models will probably perform well for the proposed reservoirs on the Burntwood River, but may not be as useful for the projects currently planned or under construction on the Nelson River with their different configuration. For example, there has been reference in past studies to the relationship between the elevation in fish mercury level and the rise in water level (e.g., Jackson 1987). Johnston et al. (1991) found this parameter accounted for a significant component of the variation in mercury level on the CRD, but rise in level should not be expected to be important in all circumstances. In the run-of-the-river reservoirs currently in operation on the Nelson River (e.g., Long Spruce and Limestone), significant increases in water level are largely contained within the natural banks of the river, with very little terrestrial organic matter being flooded. If no terrestrial organics are flooded, then the rise in water level should be of no consequence. Upstream flooding should have the dominant influence on methyl mercury availability in the reservoirs on the lower Nelson River.

8.1 Cross Lake Weir

There has been some uncertainty expressed by residents of the Cross Lake community about the potential impact of the proposed Cross Lake weir on mercury levels in fish (L. McKerness, Environment Canada, Winnipeg, MB, pers. comm.). This uncertainty seems related to the stated relationship between mercury in fish and rise in water level on the CRD. It is unlikely the resulting higher water level on Cross Lake will cause any long-term increase in fish mercury levels because the rise is contained within the natural lake basin. The weir is designed to raise the low water level without affecting the high water level. Therefore, there will be no backflooding of bog or forest terrain associated with the weir, unlike the situation along the CRD.

Experience in Sipiwesk Lake (Section 4.4) raises the question of the extent to which inundation of the macrophytes and sedges which have become established in the drawdown zone of Cross Lake will affect mercury levels in fish. Again, an elevation in fish mercury levels seems unlikely, due to important differences in shoreline composition between the lakes. The majority (92%) of the Cross Lake shoreline is bedrock controlled (Water Resources Branch 1974), which greatly limits the potential for revegetation of the drawdown zone during periods of low water. In Sipiwesk Lake, only 5% of the post-impoundment shoreline is bedrock-controlled, with the remainder comprised of flooded forest and wetland soils (Water Resources Branch 1974) which provide excellent substrate for growth of vegetation during periods of low water. Although Cross lake levels have fluctuated substantially since the Jenpeg Generating Station began operating in 1976, by as much as 3 m in a single year (Water Survey of Canada 1990), mercury levels in pike and walleye from Cross Lake are not higher than the average for lakes in northern Manitoba (0.3-0.4 $\mu g\ g^{-1}$) (Ramsey 1991). Any vegetation which becomes established in the drawdown zone apparently is not of sufficient quantity to cause a substantial elevation in net microbial methylation when it is flooded. Thus, construction of the weir should not cause increased fish mercury levels. This assessment should be confirmed by monitoring for at least 2 years after the weir is constructed.

9.0 Further Studies

The past five years of research on the mercury problem in the NFA area have answered many questions on the factors regulating microbial methylation and demethylation in new reservoirs. Our improved understanding of these processes has been applied in the development of useful models for prediction of the severity and duration of fish mercury problems in new reservoirs. This research program also has identified several questions which are in need of further investigation. These uncertainties are summarized below along with studies which might address these questions.

There is considerable circumstantial evidence which suggests the Kelsey Generating Station on the Nelson River is contributing to the mercury burden in Split Lake fish. It cannot yet be determined if this is in fact the case nor is the mechanism certain. The most probable causes are methyl mercury export from the upstream reservoir or downstream migration of contaminated fish. Another possibility is that fish in Split Lake may be feeding on fish killed in passing through the Kelsey Dam turbines.

The potential for methyl mercury export can be examined through measurements of methyl mercury in water along the Nelson River between Lake Winnipeg and Split Lake and in the other major inflows to the lake (i.e., the Aiken and Burntwood rivers). These data would indicate the location of any site of elevated methyl mercury production and determine if the methyl mercury is entering Spit Lake. The importance of fish movements from Sipiwesk Lake and the Kelsey forebay could be documented using a tagging program, as has been employed at Missi Falls on SIL. If neither of these studies finds a methyl mercury source which can explain the fish mercury levels in Split Lake, then the potential effect of fish killed in the Kelsey turbines, or the existence of a site of elevated net methylation within Split Lake should be investigated.

The numerous examples of elevated fish mercury concentrations downstream of reservoirs indicate the importance of considering downstream effects in assessments of the environmental impact of all new reservoirs. Methyl mercury can move downstream both in water and in fish. In some cases, the primary mechanism of transport, fish or water, is clear but in others there is no means of discriminating between these sources. The occurrence of much higher fish mercury levels in lakes immediately downstream from Notigi Reservoir than expected based on the extent of flooding is clear evidence of downstream transport of methyl mercury. Both mechanisms of transport probably are acting, but the relative importance of these sources remains to be determined. Fish movements out of Notigi Reservoir should be quantified to determine if they are making a significant contribution to downstream fish stocks, thereby increasing the mean mercury concentration in fish. Exports of methyl mercury in water also remain to be quantified as do the downstream extent of this transport and the mechanism by which any methyl mercury is lost from the water column. The methyl mercury survey should focus on the section of the CRD between Notigi Reservoir and Thompson as previous work indicated any methyl mercury leaving the reservoir is lost from the water column within this reach.

Studies to date have suggested that microbial demethylation rates are regulated by organic matter availability and total microbial activity. These observations need to be confirmed. There also is a need to determine why demethylation appears to be independent of sediment oxygen conditions, while methylation increases under anoxic conditions. This may provide at least a partial explanation for the differential stimulation of the methylating bacteria in flooded reservoir sediments. It is this greater stimulation of the methylators which is the cause of the elevated mercury levels in fish from reservoirs.

Concerns about the potential for elevated mercury concentrations in other food species (e.g., gull eggs, mergansers, bear), and their contribution to the mercury burden and potential health effects in humans, have been raised and need to be addressed. A pilot study should first be conducted to identify all other wild food sources, to evaluate the potential for elevated mercury levels in these sources, and to obtain an initial measure of mercury concentrations in those species for which there is the potential for elevated levels related to hydroelectric development. One criterion which might be used for selection of study species is to include all species for which fish forms part of the diet. Mercury analyses should be performed on all parts of the animals that are consumed by humans. These data can be used, along with an estimate of consumption, to determine if the animals represent a significant mercury source to consumers which may have been elevated due to flooding, and if a more extensive sampling program is warranted. This information also would be useful in assessing the potential for physiological or behavioral effects of any elevated mercury levels in these other animal groups.

There now appears to be sufficient knowledge of the cause of mercury problems in northern reservoirs to identify potential measures for the minimization or prevention of elevated fish mercury levels in new reservoirs and for the rehabilitation of fisheries in established reservoirs which continue to support elevated mercury levels. Among the proposed preventive measures, controlled burning requires an evaluation of effectiveness and an assessmegw of adverse environmental side-effects. Preliminary investigations could employ limnocorrals followed by field trials in a new reservoir if the initial results appear favourable.

Selective removal of organic matter requires knowledge on the ability of various terrestrial organic materials (plant species, soil types) to stimulate microbial methylation and the duration of this stimulation. This information could be obtained through methylation studies at selected reservoir sites with known vegetation/organic matter composition and known duration since impoundment. Unfortunately, vegetation maps were not prepared for most existing reservoirs, although a notable exception is the Complexe la Grande in northern Quebec which was extensively mapped prior to flooding.

Measures to rehabilitate fisheries which have been closed due to elevated mercury levels which might be employed on the CRD include intensive fishing, for the general reduction of mercury levels in fish, and selective fishing, to make use of species or particularly size-classes with lower mercury levels. Additional investigation is required to assess the benefits, costs, and long-term impacts on fish resources of these methods and their overall utility in the CRD.

The accumulated knowledge on the cause of reservoir mercury problems in northern Manitoba is now sufficient for the development of useful numerical models for the prediction of the severity of fish mercury problems in new reservoirs proposed for the CRD. Generalization of these models for application to projects other regions requires knowledge on the circumstances which determine the relative importance of in-reservoir and upstream effects. This requires information on the downstream extent of methyl mercury export from reservoirs and the factors affecting this distance, and some means of determining whether an upstream reservoir will have a diluting or contributing effect on methyl mercury availability in the reservoir in question. Extension of the model to other biophysical regions requires re-examination of the fish mercury/extent of flooding relationships for the relevant species. Specific field studies are required to determine the downstream extent of methyl mercury export, but the remaining issues can probably be examined in a modelling exercise using existing information.

An important part of impact assessment is the conduct of post-development studies to verify the precision and accuracy of the projection. In this regard, two seasons of post-construction monitoring of fish mercury levels in Cross Lake, using sampling methods consistent with those employed in the EMP, are suggested to confirm the prediction that the proposed weir for regulation of water levels on Cross Lake will not cause elevated fish mercury levels.

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11.0 Glossary of Technical Terms and Units of Measure

aerobic living, acting, or occurring only in the presence of oxygen.

amino acid organic compound containing both basic amino (NH_2) and acidic carboxyl (COOH) groups. Fundamental constituents of living matter, hundreds or thousands of amino acids are required to make each protein molecule.

anaerobes organisms living or occurring in the absence of free oxygen.

bacteria group of single or multi-celled, microscopic, prokaryotic organisms, lacking chlorophyll.

bioaccumulation the development of a higher concentration of a substance in an organism than in its surrounding environment.

carbanion an organic ion carrying a negative charge at a carbon position.

centrifuge a machine using centrifugal (spinning) force for separating substances of different densities or for removing moisture.

CH_3^- a methyl group which is composed of 1 carbon and 3 hydrogen atoms.

$^{14}CH_3HgI$ methyl mercuric iodide in which the carbon atom has been replaced with carbon-14, a radioactive carbon isotope.

CMMA Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion.

co-enzyme an organic compound which combines with an enzyme and plays an essential part in its catalytic reaction without being consumed in the process.

correlation the study of the simultaneous variation of two or more variables. A positive correlation is one in which one variable increases as the other increases. A negative correlation is one in which one variable decreases as the other increases.

demethylation process by which a methyl group is removed from a compound. In methyl mercury demethylation, methyl mercury is converted into elemental mercury.

DIC dissolved inorganic carbon.

electrophilic involving or having an affinity for electrons; electron-seeking.

eutrophic a type of lake with high productivity and high nutrient supply.

EMP Ecological Monitoring Program (provincial and Manitoba Hydro).

enzyme a protein which promotes chemical reactions in organisms but is not consumed by the reaction. Functions to accelerate a reaction or to permit a reaction to occur under conditions in which it would not normally proceed. Most enzymes are specific to one reaction.

FEMP Federal Ecological Monitoring Program.

Hg chemical symbol for mercury.

Hg^{2+} chemical notation for the mercuric ion.

$^{203}Hg^{2+}$ chemical notation for mercury-203, a radioisotope of mercury, occurring as a mercuric ion.

Hg^{0} chemical notation for elemental mercury, which is a silver liquid at room temperature. This is the form of mercury used in thermometers.

in situ in the natural or original environment.

ion an atom or group of atoms which carries a positive or negative charge as a result of having lost or gained one or more electrons.

isotope any of two or more species of atoms of a chemical element with the same atomic number and position in the periodic table and nearly identical chemical behaviour but with differing atomic mass or mass number and different physical properties.

isotopic equilibrium the state in which in which all isotopes of an element are completely mixed.

km kilometre (1000 metres).

limnocorral large enclosures used to isolate a water mass from a larger body of water.

loss on ignition percentage of sample weight lost after all the organic matter is combusted. A measure of organic content.

LWCN Lake Winnipeg, Churchill and Nelson Rivers.

m metre.

mercuric (ion) a divalent mercury atom with a double positive charge.

mercuric reductase an enzyme which is necessary for the conversion (reduction) of mercuric ions to elemental mercury.

mercury a naturally occurring metallic element.

methane a colourless odourless flammable gaseous hydrocarbon (CH_4) that is a product of the decomposition of organic matter.

methanogen a type of bacteria capable of forming **methane** (i.e., is methanogenic) under **anaerobic** conditions.

methanogenesis the process of producing methane.

methyl mercuric ion (CH_3Hg^+) a positively charged methyl mercury molecule.

methyl mercury term used to identify a mercury atom to which one or more methyl (CH_3) groups are attached.

methylation process by which a methyl group is attached to an atom or molecule. In mercury methylation, mercuric ions are converted to methyl mercury.

methylcobalamin chemical name for Vitamin B_{12}.

mg milligram (10^{-3} gram).

mg L^{-1} milligrams per litre.

μg L^{-1} micrograms per litre.

μg g^{-1} micrograms per gram.

microbe minute (microscopic) organism.

microbial pertaining to **microbes**.

mm millimetre (10^{-3} metre).

M/D the ratio of specific methylation to specific demethylation rates. Also called the **methylation balance**.

methylation balance M/D ratio. A measure of the net rate of methyl mercury production by bacteria.

monomethyl mercury an organic form of mercury containing one methyl (CH_3) group.

$NADP^+$ nicotinamide adenine dinucleotide phosphate (oxidized form). Also called Co-enzyme II. An oxidizing-reducing co-enzyme involved in synthesis of metabolic products which is not consumed in the process.

NADPH nicotinamide adenine dinucleotide phosphate (reduced form).

net-plankton **plankton** organisms large enough to be captured in a mesh net of a specified size.

NFA Northern Flood Agreement.

ng nanogram (10^{-9} gram).

ng L^{-1} nanograms per litre.

ng g^{-1} nanograms per gram.

non-enzymatic not requiring an **enzyme**.

obligate biologically essential for survival.

oligotrophic a type of lake with low productivity and low nutrient supply.

organomercurial lyase an enzyme which is necessary for the removal of the methyl group from methyl mercury

photolysis chemical decomposition by the action of radiant energy (e.g., sunlight).

plankton small animals and plants which float or drift in lakes.

ppm a concentration expressed in parts per million. Equivalent units are $\mu g\ g^{-1}$ and $mg\ L^{-1}$.

primary productivity the total amount of organic matter synthesized by the autotrophic (mainly plants) organisms of na ecosystem.

protein a complex organic compound composed of many **amino acids**.

radioactive the property of having atoms that break up spontaneously and send out radiation.

radioisotope a **radioactive** isotope (form) of an element.

SIL Southern Indian Lake

sulphate reducers a group of bacteria which convert sulphate (SO_4) to hydrogen sulphide (H_2S; a gas which smells like rotten eggs) under anoxic conditions.

sulfhydryl group a chemical functional (SH) group, comprised of one sulphur and one hydrogen atom, occurring in the **amino acid** cysteine.

supernatant the liquid overlying material deposited by centrifugation.

Vitamin B_{12} a cobalt-containing vitamin required by many organisms.

EFFECTS OF THE CHURCHILL RIVER DIVERSION ON WATERFOWL USE OF THE RAT-BURNTWOOD RIVERS SYSTEM AND ON THE NELSON HOUSE BAND

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ABSTRACT

Surveys carried out in September 1988 indicated that use of the Rat-Burntwood rivers system by ducks in fall declined following completion of the Churchill River Diversion project. Duck numbers overall in 1988 were 95 percent lower than recorded during 1973 surveys. Dabbling ducks accounted for most of the decline (97 percent reduction), while diving duck numbers fell by 89 percent. Survey results were insufficient to assess levels of use by Canada geese with any degree of confidence.

The observed reduction in duck use of the Rat-Burntwood area cannot be attributed to a regional decline in populations, since no dramatic decline was evident in regional survey data (Stratum 24). It is more likely that the reduction was associated with a 4 to 5 m increase in water levels that removed marshy cover along shorelines, leaving behind poor quality habitat characterized by dead trees, fallen timber and other debris, and eroding banks. However, the limited number of survey years (1 pre- and 1 post-diversion) and the variability inherent in aerial waterfowl counts make it impossible to state, with certainty, the cause and significance of the changes observed.

Residents of Nelson House interviewed during the study indicated that low duck numbers, high water levels, and debris along flooded shores have reduced opportunities for Band members to hunt and view waterfowl.

TABLE OF CONTENTS

Page

ABSTRACT

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION 1

2. STUDY AREA 3

3. CHURCHILL RIVER DIVERSION PROJECT 5

4. PRE-PROJECT ENVIRONMENTAL ASSESSMENT 8

- 4.1 Study Findings 8
- 4.2 Study Board Recommendations 9

5. METHODS 10

- 5.1 Impact Assessment and Impact Management Approach 10
  - 5.1.1 Biophysical Impacts 12
  - 5.1.2 Socio-Economic Cultural Impacts 12
  - 5.1.3 Impact Management 13
- 5.2 Data Collection 13
  - 5.2.1 Effects on Waterfowl 13
  - 5.2.2 Effects on Nelson House Band 14

6. RESULTS 17

- 6.1 Waterfowl Use 17
- 6.2 Hunter Interviews 23

TABLE OF CONTENTS (Cont'd) **Page**

7. DISCUSSION 25

    7.1 Pre- and Post-Diversion Waterfowl Use 25

    7.2 Possible Causes of Waterfowl Use Changes 31

        7.2.1 Regional Waterfowl Population Changes 34
        7.2.2 Modifications to Habitat 41

8. SUMMARY AND CONCLUSIONS 47

    8.1 Project Impacts on Waterfowl 47

    8.2 Project Impacts on the Nelson House Band 47

9. LITERATURE CITED 48

APPENDIX

TABLES

Page

1. Lengths (km) of post-diversion surveys of the Rat-Burntwood river system 16

2. Numbers and densities of dabbling ducks observed during post-diversion surveys of the Rat-Burntwood rivers area. 19

3. Numbers and densities of diving ducks observed during post-diversion surveys of the Rat-Burntwood rivers area. 19

4. Numbers and densities of ducks observed during post-diversion surveys of the Rat-Burntwood rivers area. 22

5. Numbers and densities of Canada geese observed during post-diversion surveys of the Rat-Burntwood rivers area. 22

6. Numbers of dabbling ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area. 26

7. Numbers of diving ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area. 26

8. Numbers of ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area. 27

9. Numbers of Canada geese observed during pre-diversion surveys of the Rat-Burntwood rivers area. 27

10. Changes in waterfowl numbers in the Rat-Burntwood rivers area from pre-diversion to post-diversion based on highest survey counts. 28

11. Summary of changes in waterfowl use of the Rat-Burntwood for lakes represented by both pre- and post-diversion counts. 32

TABLES (Cont'd)

Page

12. Adjusted population estimates (thousands) for dabblers, divers and Canada geese in stratum 24 during pre-diversion (1965-75) and post-diversion (1978-88) periods (from U.S. Fish & Wildlife Service). 36

13. Changes in regional (stratum 24) and local (Rat-Burntwood) waterfowl use from pre-diversion (1973) to post-diversion (1988). 39

14. Pre- and post-diversion water levels (m a.s.l.) at Footprint Lake, Nelson House hydrometric station (#05TF001). 42

FIGURES

Page

1. The Rat-Burntwood rivers system. 4

2. Area affected by the Churchill River Diversion project. 7

3. The impact assessment and impact management approach used in the study. 11

4. Rat-Burntwood river system: waterfowl survey route, fall 1988. 15

5. Dabbling duck use of the Rat-Burntwood area in fall, 1988. 18

6. Diving duck use of the Rat-Burntwood area in fall, 1988. 18

7. Duck use of the Rat-Burntwood area in fall, 1988. 21

8. Canada goose use of the Rat-Burntwood area in fall, 1988. 21

9. Waterfowl-related concerns and adverse effects identified during hunter interviews at Nelson House. 24

10. Pre- and post-diversion use of Wapisu Lake area by waterfowl based on highest survey counts. 29

11. Pre- and post-diversion use of Rat River area by waterfowl based on the highest survey counts. 29

12. Pre- and post-diversion use of Threepoint Lake area by waterfowl based on highest survey counts. 30

13. Pre- and post-diversion use of lower Burntwood River area by waterfowl based on highest survey counts. 30

14. Pre- and post diversion use of Wuskwatim Lake area by waterfowl based on highest survey counts. 33

15. Pre- and post diversion use of Rat-Burntwood area by waterfowl based on highest survey counts. 33

FIGURES (Cont'd)

Page

16. Location of waterfowl survey stratum 24. 35

17. Adjusted population estimates of dabblers and divers in stratum 24 (northern Manitoba) 1955-1988. 37

18. Estimates of indicated Canada goose pairs in stratum 24 (northern Manitoba) 1955-1988, and the estimated fall flight of Canada Geese, Eastern Prairie Population, 1967-1988, based on production survey data. 37

19. Pre- and post-diversion water levels at Footprint Lake (Nelson House) hydrometric station (#05TF001). 43

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1.0 INTRODUCTION

In the early 1970's, Manitoba Hydro began to proceed with developments to harness the hydroelectric potential of the Nelson River. This involved the diversion of a major portion of the flow of the lower Churchill River into the Nelson River, referred to as the Churchill River Diversion (CRD) project, and regulation of the outflow from Lake Winnipeg through the Lake Winnipeg Regulation project. The CRD project is the subject of this report and is described in **3.0 CHURCHILL RIVER DIVERSION PROJECT**.

In response to concerns expressed about the effects of this development scheme on the natural environment and on northern communities that depended on natural resources, the governments of Canada and Manitoba initiated the Lake Winnipeg, Churchill and Nelson Rivers Study. In April 1975, a report was issued by the Lake Winnipeg, Churchill and Nelson Rivers Study Board (Study Board or LWCNRSB) which described the environmental and social impacts that were expected to occur as a result of the hydroelectric development. The report also recommended measures to minimize or offset impacts, and specified additional studies that should be carried out. Study Board findings and recommendations relevant to the present study are discussed in **4.0 PRE-PROJECT ENVIRONMENTAL ASSESSMENT**.

To ensure that all members of Indian communities affected by the hydroelectric development would be adequately compensated, the **Northern Flood Agreement** (**NFA**) was entered into on December 16, 1977 by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee. The Northern Flood Committee was established to represent the interests of the Nelson House, Norway House, Cross Lake, Split Lake and York Factory bands. Included in the **NFA** is the requirement that the adverse effects of the development be monitored to determine how Indian Band members have been affected and how they should be compensated.

The **NFA** also established an arbitration procedure whereby injured parties could file claims and obtain suitable remedies for damages suffered. One such claim, Claim 18, lead to the initiation of the Federal Ecological Monitoring Program (FEMP) in the spring of 1986. FEMP was to address federal responsibilities under the NFA with respect to environmental research and monitoring. Claim 18 and various provisions of the NFA are described in **APPENDIX A, NORTHERN FLOOD AGREEMENT**.

Funds provided through FEMP enabled the present study to be initiated in response to concerns expressed by Nelson House Band that opportunities to hunt waterfowl in the fall had declined following completion of the CRD project.

2.0 STUDY AREA

In the context of this study, the Rat-Burntwood rivers system extends from the western end of Wapisu Lake, located downstream from the Notigi Control Structure, to the southern end of Wuskwatim Lake (Figure 1). The study area includes Wapisu, Threepoint, Footprint, Wuskwatim and Honeymoon lakes, reaches of the Rat and Burntwood rivers connecting these lakes, and the section of the Burntwood River between Threepoint Lake and Hopover Rapids. The community of Nelson House is located on Footprint Lake. Notakikwaywin Lake, located approximately 15 km north of Nelson House and not connected to the Rat-Burntwood rivers system, was included in the study area for comparison purposes.

The topography of the area reflects the underlying bedrock and the area's geomorphic history, particularly that of glacial and post-glacial periods (LWCNRSB 1975a). The distribution of surface material has been influenced by glaciation and glacial Lake Agassiz. Organic deposits, glacial till, lacustrine clays and glaciofluvial sands comprise the surficial material. Permafrost is common, especially if surficial materials are fine and drainage is poor. Organic deposits are widespread, consisting mainly of various types of peat (LWCNRSB 1975b).

The shorelines of Wapisu, Threepoint and Wuskwatim lakes are bedrock-controlled to a large extent. Prior to the diversion, a thin fringe of sedge (*Carex* spp.) and willow (*Salix* spp.) occurred along the shore of Wapisu Lake. Marsh areas were found at the shallow end of embayments and at the outlet of the Lake. Threepoint Lake used to be quite shallow before the diversion, and pondweeds (*Potomogeton* spp.) were present throughout. Marshes occurred in small bays at the mouth of small tributaries; beaches were common. A wide low-lying area along the Burntwood River upstream of Wuskwatim Lake used to contain extensive marsh habitat prior to the diversion. Submerged aquatic plant growth was so dense in mid-summer that navigation was restricted to a narrow section of the central channel. Further upstream, the river flows through a series of rapids contained within elevated banks. Before diversion, about one-half of the shoreline of Wuskwatim Lake was bedrock controlled with low alluvium and marshy sites comprising the remainder. Extensive marsh habitat occurred at the south end of the Lake before diversion (LWCNRSB 1975a,b). Kellerhals Engineering Services

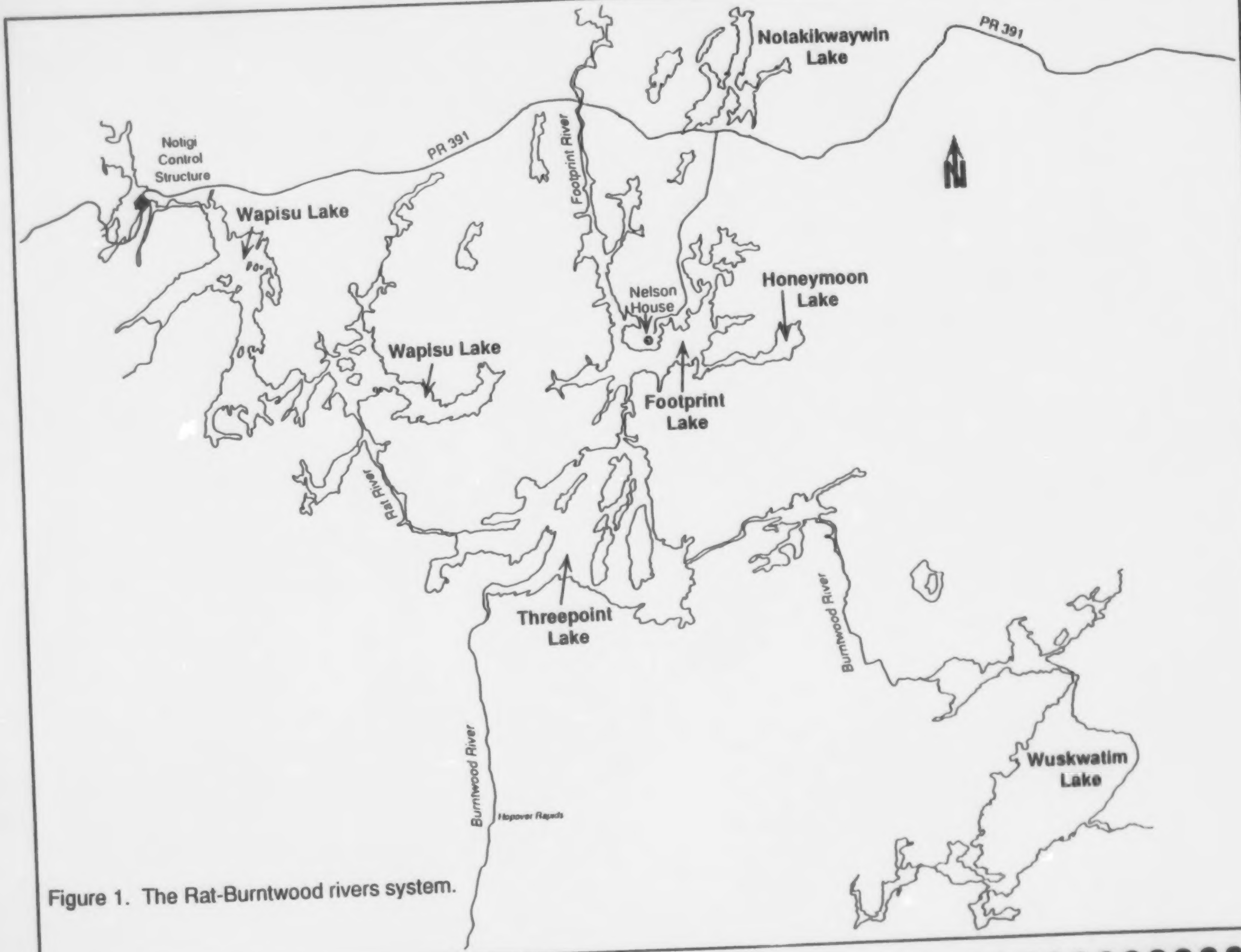
PR 391
Notigi Control Structure
PR 391
Footprint River
Notakikwaywin Lake
Wapisu Lake
Wapisu Lake
Nelson House
Honeymoon Lake
Footprint Lake
Rat River
Threepoint Lake
Burntwood River
Burntwood River
Hopover Rapids
Wuskwatim Lake
N

Figure 1. The Rat-Burntwood rivers system.

Ltd. (1988) noted that shorelines at the southern end of the lake are eroding at a rate of .2 m per year.

Black spruce (*Picea mariana*), white spruce (*Picea glauca*), trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*) and white birch (*Betula papyrifera*) dominate most forest communities in the area. Willow, alder (*Alnus* spp.) and sedge communities occur on wet or saturated sites. Before diversion, marsh vegetation was characterized by a band of sedge along shoreline areas with an outer band of bulrush (*Scirpus* spp.) in deeper water. Floating aquatics, such as duckweed (*Lemna* spp.), were common in shallow sheltered areas, and pondweed (*Potomogeton* spp.) often occurred in deeper water beyond the bulrush margin (LWCNRSB 1975b). With the increased water levels associated with the diversion, marsh areas have become inundated so that shorelines today are obstructed by debris and dead standing trees. For example, Kellerhals Engineering Services Ltd. (1988) estimated that, at Footprint and Honeymoon lakes, the length of shoreline containing standing debris had increased from 0.2 km in 1972 to 56 km in 1985.

The climate is continental and is characterized by short, cool summers and long, cold winters. Annual precipitation averages 45 cm, 60 percent of which falls as rain between April and October (LWCNRSB 1975b).

3.0 CHURCHILL RIVER DIVERSION PROJECT

The following details of the CRD Project were extracted from the Summary Report of the Lake Winnipeg, Churchill and Nelson Rivers Study (LWCNRSB 1975c), supplemented by other information from Manitoba Hydro.

The Churchill River Diversion is a major component of hydroelectric development on the Nelson River. In simple terms, its purpose in the overall scheme is to divert approximately 75 percent of the flows of the lower Churchill River into the Nelson River drainage to augment Nelson River flows and maximize the production of electric power at generating stations located along that river.

A control structure was installed at Missi Falls where flows from Southern Indian Lake historically entered the lower Churchill River (Figure 2). The control structure permitted natural outflows to be held back in Southern Indian Lake and caused the mean level of the lake to increase by about three metres. The rise in water level enabled water to be diverted from Southern Indian Lake at South Bay and into the Rat River located in the Nelson River drainage basin. The water flows by gravity from South Bay to Issett Lake on the Rat River through an excavated channel. Control of the flow into the Rat-Burntwood rivers system is achieved by a control structure located at the outlet of Notigi Lake.

Construction of the Missi Falls and Notigi control structures began in February 1973 and September 1973, respectively. Both structures were completed by August 1976. Connection of Southern Indian Lake to the Rat River occurred on June 2, 1976, with breaching of the South Bay cofferdam. The diversion was phased in over 1976 and 1977.

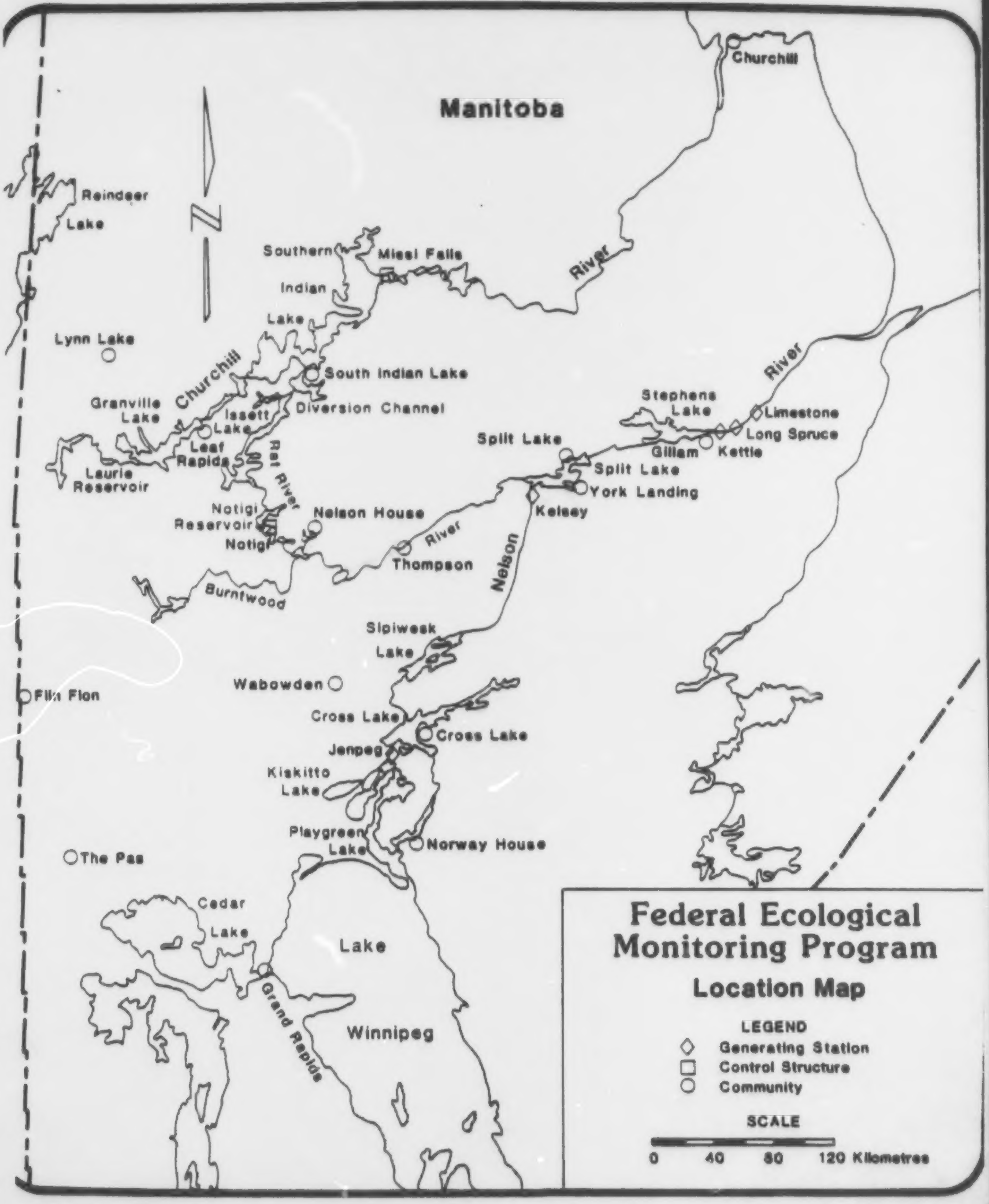
Manitoba
N
Churchill
Reindeer Lake
Southern Indian Lake
Missi Falls
River
Lynn Lake
Churchill
South Indian Lake
Granville Lake
Issett Lake
Diversion Channel
Leaf Rapids
Laurie Reservoir
Rat River
Notigi Reservoir
Notigi
Nelson House
Burntwood
River
Thompson
Nelson
Split Lake
Split Lake
York Landing
Kelsey
Stephens Lake
Gillam
Kettle
Long Spruce
Limestone
River
Sipiwesk Lake
Wabowden
Flin Flon
Cross Lake
Cross Lake
Jenpeg
Kiskitto Lake
Playgreen Lake
Norway House
The Pas
Cedar Lake
Grand Rapids
Lake Winnipeg
Federal Ecological Monitoring Program
Location Map
LEGEND
◇ Generating Station
☐ Control Structure
○ Community
SCALE
0
40
80
120 Kilometres

Figure 2. Area affected by the Churchill River Diversion Project.

4.0 PRE-PROJECT ENVIRONMENTAL ASSESSMENT

4.1 Study Findings

Following its investigation into the potential physical impacts of the Churchill River Diversion, the LWCNRSB (1975a) concluded the following:

> *"After the diversion of flows from the Churchill River and regulation at the Notigi control structure, the mean discharge in the Burntwood River will be increased nine to ten-fold. The impact on lakeshores underlain by bedrock will be one of erosion until bedrock is reached. Such shorelines will remain indefinitely cluttered with fallen trees. Embayments and other protection shorelines will remain congested with standing dead trees. Sand beaches and marshes will be inundated and will not redevelop to the extent they were in the natural state. Steep clay banks will continue to erode, although at an accelerated rate. These shorelines will likely appear much as they did in the natural state. Steep bedrock shorelines will not be significantly altered."*
>
> *"Each shoreline type will be affected differently. Little change will occur where bedrock controls the channel width but clay shores may recede approximately 60 to 70 ft. on either side. The main difference between riverbanks and lakeshores is the removal of flooded and fallen trees. Riverbanks will not likely contain the amount of dead standing trees that will typify protected lakeshores."*

Webb and Foster (1974) assessed the impact of the CRD project on waterfowl and waterfowl habitat. Concerning the effect of the diversion on fall waterfowl use of the Rat-Burntwood rivers system, the following predictions were made:

> *"Features which make certain areas along the Rat and Burntwood Rivers attractive to staging waterfowl will be lost after flooding; these include the large expanses of sedge and exposed mud flats. Food on the latter areas, both plant and animal, is the critical factor in attracting ducks and geese.*

> *Large expanses of potentially productive low land are being flooded throughout the Diversion route. The time required for aquatic vegetation to establish on these areas will determine future attractiveness to waterfowl. Clearing along shorelines in these areas would greatly speed reestablishment of attractive habitat.*
>
> *Waterfowl using the large, open lake areas for shore migration stops will not be significantly affected by flooding."*

4.2 Study Board Recommendations

The Study Board made a number of general recommendations that concerned the entire Nelson River hydroelectric development scheme, and others that were specific to certain issues or geographic areas.

Recommendations of general application that are relevant to the present study are:

> *5. That a mechanism be established to deal with social and related economic issues including:*
>
> > *(c) monitoring and analysis of ongoing social and economic changes related to hydroelectric development and more generally northern development.*
>
> *10. That appropriate government departments and agencies develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the affected area.*

No specific recommendations were made in relation to the impacts of the CRD project on waterfowl use of the Rat-Burntwood area.

5.0 METHODS

5.1 Impact Assessment and Impact Management Approach

The flowchart in Figure 3 illustrates the approach used in determining how waterfowl-related interests and opportunities of Nelson House Band Indians in fall have been affected by the CRD project and, therefore, what compensation is required by the NFA. References are made in the flowchart to the pertinent recommendations of the Study Board and provisions of the NFA which are addressed by the process. Reference to "adverse" effects or impacts in this report reflects terminology of the NFA and does not represent any pre-conclusion of this study.

The assessment process begins with identifying the effects of the CRD project on waterfowl use of the Rat-Burntwood area in fall (biophysical impacts). The direct, indirect, or cumulative consequences of biophysical impacts are expected to affect the economic, cultural, and social life of Nelson House people.

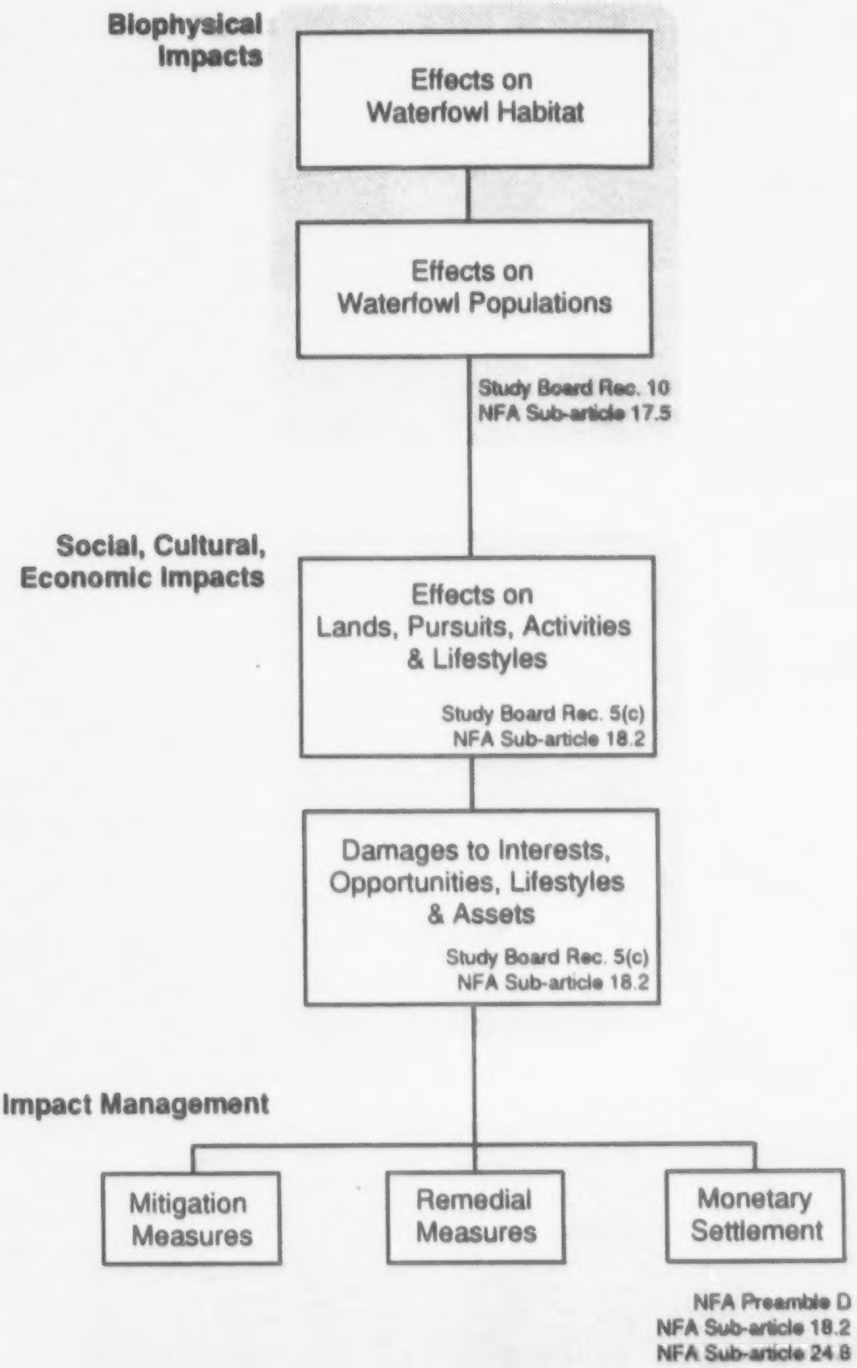
Biophysical Impacts
Effects on Waterfowl Habitat
Effects on Waterfowl Populations
Study Board Rec. 10
NFA Sub-article 17.5
Social, Cultural, Economic Impacts
Effects on Lands, Pursuits, Activities & Lifestyles
Study Board Rec. 5(c)
NFA Sub-article 18.2
Damages to Interests, Opportunities, Lifestyles & Assets
Study Board Rec. 5(c)
NFA Sub-article 18.2
Impact Management
Mitigation Measures
Remedial Measures
Monetary Settlement
NFA Preamble D
NFA Sub-article 18.2
NFA Sub-article 24.8

Figure 3. The impact assessment and impact management approach used in the study.

5.1.1 Biophysical Impacts

Determining how the CRD project has affected waterfowl use of the Rat-Burntwood area partially fulfills Recommendation 10 of the Study Board and **NFA** sub-article 17.5 (see Appendix A).

Recommendation 10 recognizes that ongoing management of the Rat-Burntwood area requires that information first be gathered on how the area has been impacted by development. The present study focuses on how use of the Rat-Burntwood area by fall-staging waterfowl has been affected by the CRD project.

NFA sub-article 17.5 requires that monitoring of effects of the CRD project be carried out to "provide such information as may be necessary to give effect" to the **NFA** (see Appendix A). Giving effect to the **NFA** means ensuring that all members of the five **NFA** bands, who have been or may be adversely affected by the hydroelectric development, are dealt with or compensated "fairly and equitably" (**NFA** Preamble D).

Collection of data that document the effects of CRD on waterfowl use of the Rat-Burntwood area constitutes compliance with sub-article 17.5 with respect to the waterfowl-related impacts that the Nelson House Band claims to have experienced. In addition, documentation of effects on waterfowl in the Rat-Burntwood area represents a response, in part, to Part III of Claim 18, which embodies the intent and meaning of Recommendation 10 and **NFA** sub-article 17.5. Claim 18 is described in Appendix A.

5.1.2 Socio-Economic-Cultural Impacts

Determining how CRD has affected the waterfowl-related interests of Nelson House Band people partially fulfills Study Board Recommendation 5(c) and **NFA** sub-article 18.2.

Recommendation 5(c) called for "monitoring and analysis of ongoing social and economic changes related to hydroelectric development". The present study addresses those changes experienced by Nelson House Band that are associated with effects on waterfowl use of the Rat-Burntwood area, i.e., changes in waterfowl hunting opportunities.

Collection of information that documents the effects of CRD on the waterfowl hunting opportunities of Nelson House Band constitutes compliance with **NFA** sub-article 18.2 with respect to the waterfowl-related impacts experienced by the Nelson House Band. In addition, this documentation represents a response, in part, to Part II of Claim 18 which embodies the intent and meaning of Recommendation 5(c) and **NFA** sub-article 18.2. Finally, information dealing with adverse effects on waterfowl hunting opportunities allows damages to the waterfowl-related interests, opportunities, lifestyles and assets of Nelson House Band Indians to be defined.

5.1.3 Impact Management

Impact Management refers to a mechanism, or set of mechanisms, by which the impacts of a project may be managed, whether they be biophysical (e.g., directly affecting waterfowl) or socio-economic-cultural (e.g., affecting waterfowl hunting opportunities). Suitable and adequate compensation for adverse effects on **NFA** bands would constitute redress for damages suffered by the bands. **NFA** sub-article 24.8 refers to three forms of compensation or redress for damages suffered: "mitigating measures, remedial measures and monetary settlement."

5.2 Data Collection

5.2.1 Effects on Waterfowl

Fall staging waterfowl use of the Rat-Burntwood area following diversion was determined through a series of low-altitude aerial surveys carried out in 1988. The surveys were conducted using a Bell 206B helicopter on September 1, 9 and 26. Altitudes of between 30 and 40 m above ground or water level were maintained during the surveys and the helicopter was positioned to enable the single observer to note the locations of waterfowl distributed along the shoreline.

Survey routes followed shorelines of Wapisu, Threepoint, Wuskatim and Honeymoon lakes and the Rat and Burntwood rivers (Figure 4). A total of 181.6 km were surveyed on September 1, 169.1 km on September 9, and 196.4 km on September 26 (Table 1). In addition, on September 26, 14.2 km of Notakikwaywin Lake shoreline were surveyed to determine whether the level of waterfowl use of areas located on the Rat-Burntwood rivers system and off the system were similar. Notakikwaywin Lake is located approximately 15 km north of Nelson House (Figure 4).

5.2.2 Effects on Nelson House Band

On February 19, 1988, one-on-one interviews were held with seven waterfowl hunters at Nelson House. Hunters were asked to describe their waterfowl hunting experiences before and after the Churchill River diversion, traditional hunting locations prior to diversion, hunting locations following the diversion, hunting success at traditional hunting locations before and after the diversion, and species of waterfowl hunted. They were also asked to share their observations on changes in numbers and distribution of waterfowl and their perceptions on the causes of these changes and the influence of the Churchill River Diversion. Because the purpose of the study was to investigate the effects of the diversion on fall waterfowl use and fall hunting, discussions focussed on the fall hunting period.

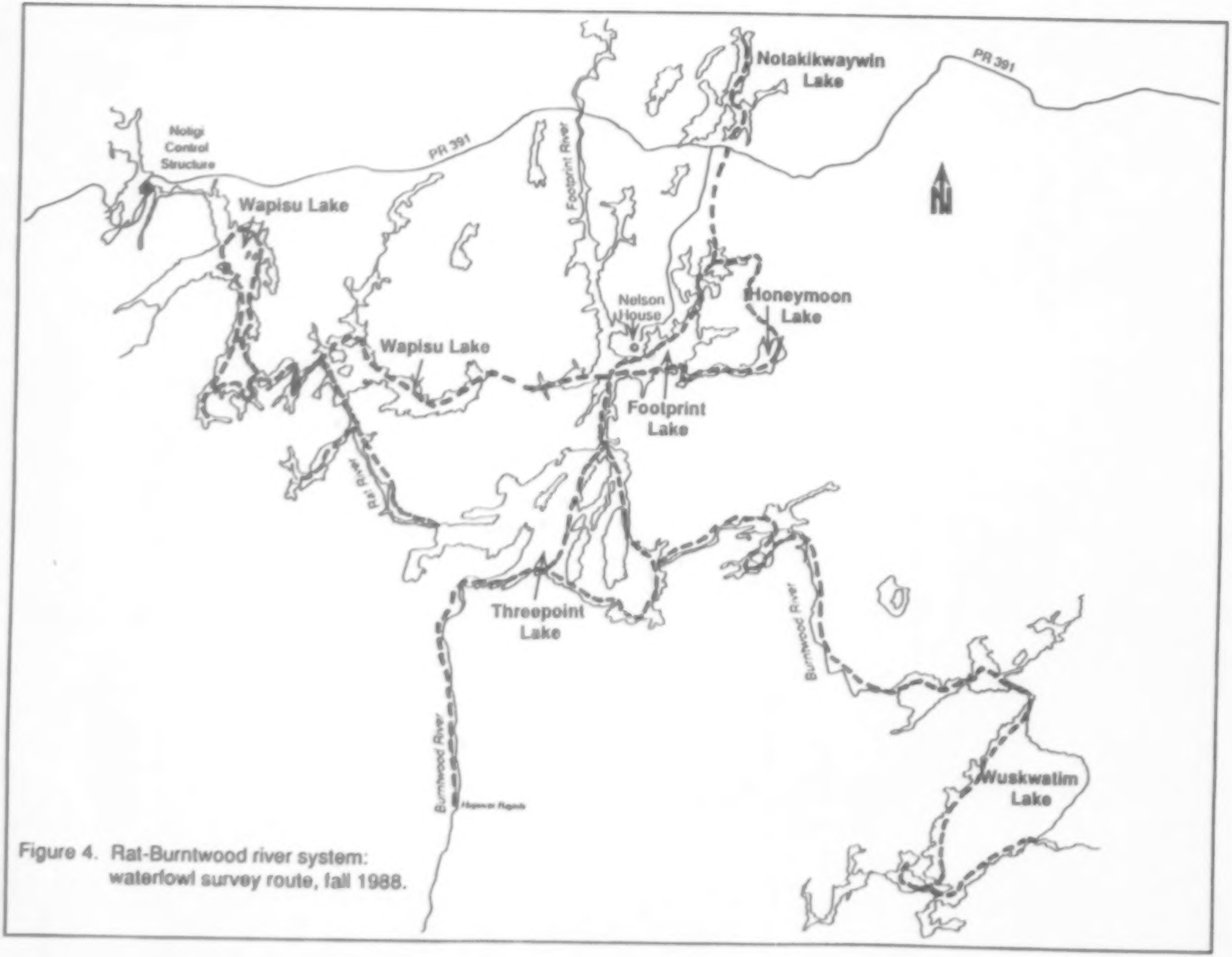
Notigi Control Structure
PR 391
Wapisu Lake
Wapisu Lake
Footprint River
Nelson House
Notakikwaywin Lake
PR 391
N
Honeymoon Lake
Footprint Lake
Rat River
Threepoint Lake
Burntwood River
Burntwood River
Wuskwatim Lake

Figure 4. Rat-Burntwood river system: waterfowl survey route, fall 1988.

Table 1. Lengths (km) of post-diversion surveys of the Rat-Burntwood river system.

Area	Sept. 1, 1988	Sept. 9, 1988	Sept. 26, 1988
Wapisu Lake	59.2	38.4	58.6
Rat River	20.0	20.0	20.0
Burntwood River (upstream)	17.6	17.6	17.6
Threepoint Lake	9.0	9.0	16.1
Burntwood River (downstream)	32.2	32.2	32.2
Wuskwatim Lake	34.8	43.1	43.1
Honeymoon Lake	8.8	8.8	8.8
Notakikwaywin Lake	n.s.	n.s.	14.2
TOTAL*	181.6	169.1	196.4

* Does not include Notakikwaywin Lake.
n.s. = not surveyed

6.0 RESULTS

6.1 Waterfowl Use

Data on waterfowl use have been summarized for the following survey areas: Wapisu Lake, Rat River, Burntwood River upstream of Threepoint Lake, Threepoint Lake, Burntwood River downstream of Threepoint Lake, Wuskwatim Lake, Honeymoon Lake, and Notakikwaywin Lake (Figure 1). Where applicable, boundaries between areas were chosen to coincide with those of Webb and Foster's (1974) pre-diversion study to enable comparison of pre-and post-diversion counts. All survey areas except Notakikwaywin Lake are located along the Rat-Burntwood rivers system and are affected by the CRD project. Notakikwaywin Lake lies just north of Provincial Road 391, approximately 15 km northeast of Nelson House, and is not influenced by the CRD project.

Date presented here are summarized for dabbling ducks (Anatinae), diving ducks (primarily Aythyinae), and Canada Geese (Branta canadensis).

Highest densities of dabbling ducks observed per survey segment of the Rat-Burntwood system in fall 1988 (Figure 5, Table 2) ranged from 0.02 birds/km (Wuskwatim Lake) to 2.51 birds/km (Honeymoon Lake). No dabblers were seen on Threepoint Lake during any of the three surveys. The largest number of dabblers observed on a given survey was 106 birds (0.63 ducks/km) on September 9. The 2.47 dabblers/km recorded at Notakikwaywin Lake on September 26 was 7.3 times greater than the highest density (0.34 birds/km) observed on the Rat-Burntwood system on that date (Honeymoon Lake).

Maximum numbers of diving ducks ranged from 0.03 birds/km on the downstream segment of Burntwood River to 1.04 birds/km on Honeymoon Lake (Figure 6, Table 3). The largest number of divers observed on a single survey was 35 (0.21 birds/km) on September 9. Twenty-seven (27) divers (1.91/km) were recorded on Notakikwaywin Lake on September 26, the highest same-day density recorded on the Rat-Burntwood system was 0.23 diver/km (Honeymoon Lake).

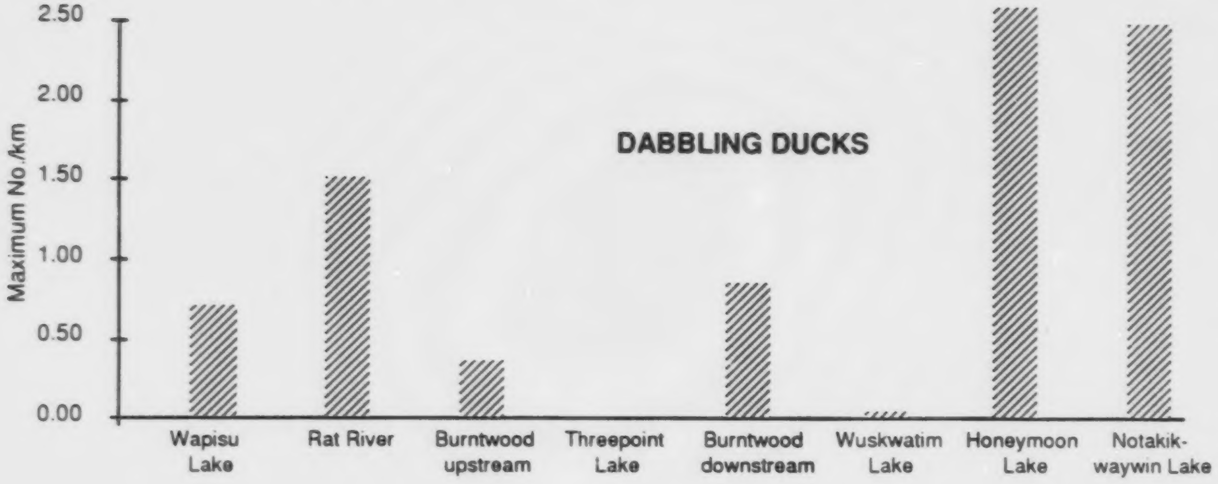

Figure 5. Dabbling duck use of the Rat-Burntwood area in Fall, 1988

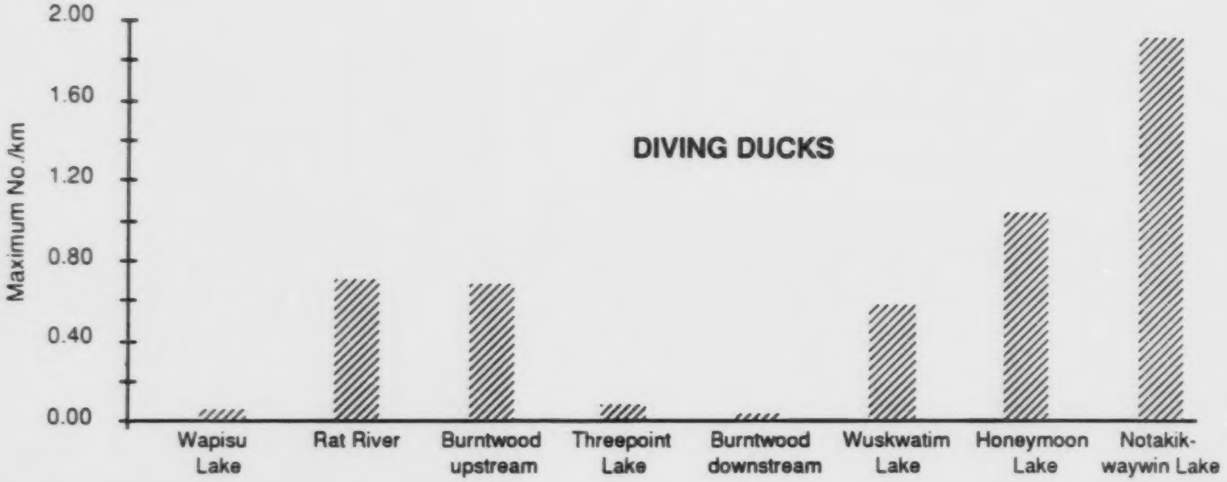

Figure 6. Diving duck use of the Rat-Burntwood area in Fall, 1988

Table 2. Numbers and densities* of dabbling ducks observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	0		25	0.65	10	0.17	25	0.65
Rat River	4	0.20	30	1.50	0		30	1.50
Burntwood River (upstream)	5	0.28	4	0.23	6	0.34	6	0.34
Threepoint Lake	0		0		0		0	0.00
Burntwood River (downstream)	1	0.03	25	0.78	0		25	0.78
Wuskwatim Lake	0		0		1	0.02	1	0.02
Honeymoon Lake	2	0.30	22	2.51	3	0.34	22	2.51
Notakikwaywin Lake	n.s.		n.s.		35	2.47	35	2.47
TOTAL & AVG. DENSITY**	**12**	**0.07^**	**106**	**0.63^**	**20**	**0.1^**	**144**	

* birds/km

** Excludes Notakikwaywin Lake

^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)

n.s. = not surveyed

Table 3. Numbers and densities* of diving ducks observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	1	0.02	2	0.05	2	0.03	2	0.05
Rat River	0		14	0.70	0		14	0.70
Burntwood River (upstream)	0		12	0.68	0		12	0.68
Threepoint Lake	0		0		1	0.06	1	0.06
Burntwood River (downstream)	0		1	0.03	0		1	0.03
Wuskwatim Lake	20	0.58	5	0.12	5	0.12	20	0.58
Honeymoon Lake	7	1.04	1	0.11	2	0.23	7	1.04
Notakikwaywin Lake	n.s.		n.s.		27	1.91	27	1.91
TOTAL & AVG. DENSITY**	**28**	**0.15^**	**35**	**0.21^**	**10****	**0.05^**	**84**	

* birds/km

** Excludes Notakikwaywin Lake

^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)

n.s. = not surveyed

Maximum numbers densities of dabblers, divers and unidentified ducks combined as shown in Figure 7 and Table 4, varied from 0.55/km (Threepoint Lake, September 1) to 2.63/km (Honeymoon Lake, September 9). The largest number of ducks (141; 0.83/km) was observed on September 9. The September 26 survey of Notakikwaywin Lake yielded 4.38 ducks/km, compared to the maximum of 0.57/km recorded on the same day in the Rat-Burntwood river system (Honeymoon Lake).

Canada geese were observed along only two survey segments: Rat River and Wuskwatim Lake (Figure 8, Table 5). Most of the 75 geese (1.74/km) were observed upstream of Wuskwatim Lake on September 9. No geese were observed at Notakikwaywin Lake.

In summary, low numbers of ducks were observed on the Rat-Burntwood system during post-diversion surveys. Densities were considerably higher during the single survey of Notakikwaywin Lake, which was not affected by the diversion. Canada goose numbers were low in all surveyed areas.

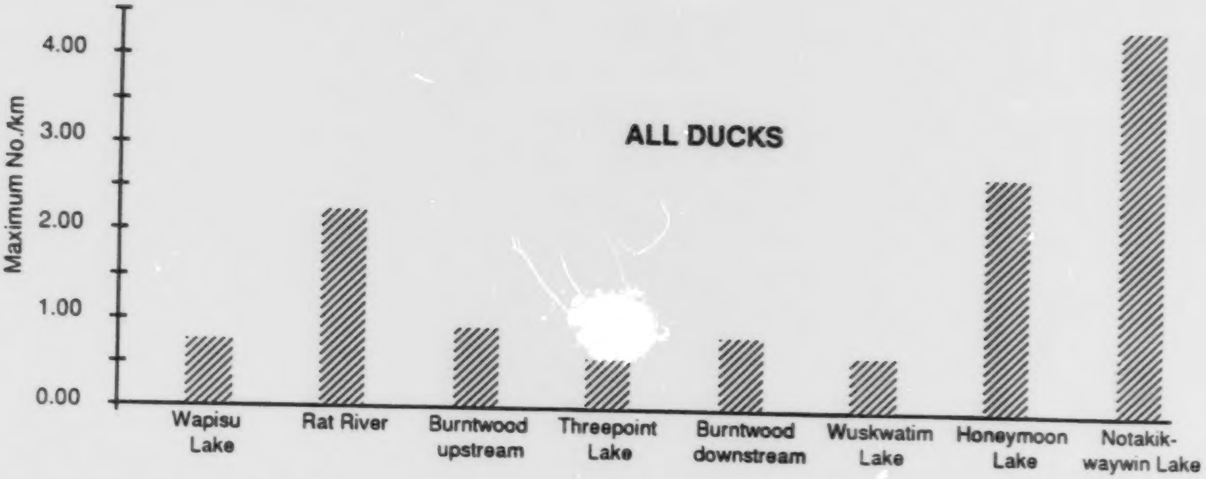

Figure 7. Duck use of the Rat-Burntwood area in Fall, 1988

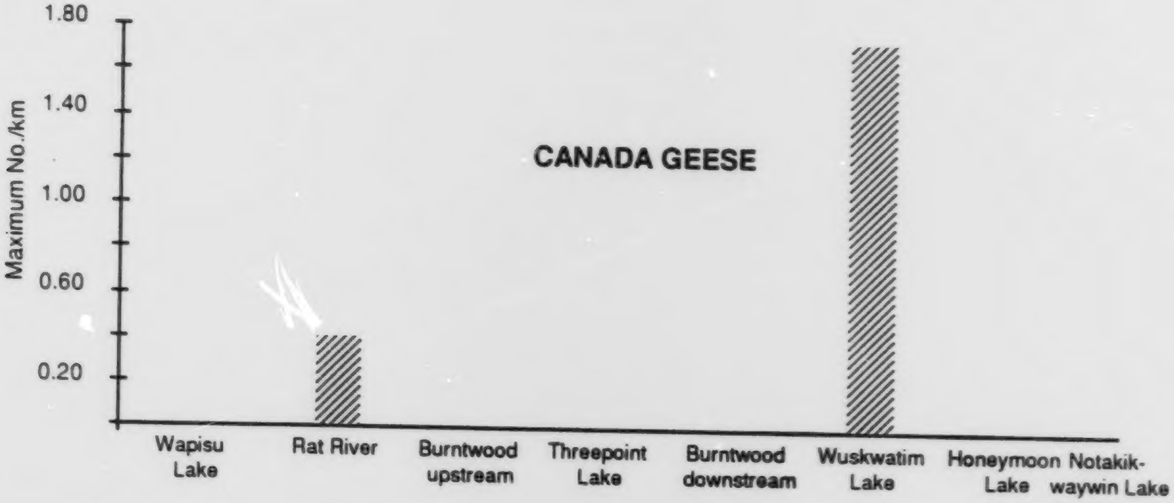

Figure 8. Canada goose use of the Rat-Burntwood area in Fall, 1988

Table 4. Numbers and densities* of ducks observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	1	0.02	27	0.70	12	0.20	27	0.70
Rat River	4	0.20	44	2.20	0		44	2.20
Burntwood River (upstream)	5	0.28	16	0.91	6	0.34	16	0.91
Threepoint Lake	5	0.55	0		1	0.06	5	0.55
Burntwood River (downstream)	1	0.03	26	0.81	0		26	0.81
Wuskwatim Lake	20	0.58	5	0.12	6	0.14	20	0.58
Honeymoon Lake	9	1.33	23	2.63	5	0.57	23	2.63
Notakikwaywin Lake	n.s.		n.s.		62	4.38	62	4.38
TOTAL & AVG. DENSITY**	45	0.25^	141	0.83^	30**	0.15^	223	

* birds/km

** Excludes Notakikwaywin Lake

^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)

n.s. = not surveyed

Table 5. Numbers and densities* of Canada geese observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	0		0		0		0	0.00
Rat River	8	0.40	2	0.10	0		8	0.40
Burntwood River (upstream)	0		0		0		0	0.00
Threepoint Lake	0		0		0		0	0.00
Burntwood River (downstream)	0		0		0		0	0.00
Wuskwatim Lake	0		75	1.74	0		75	1.74
Honeymoon Lake	0		0		0		0	0.00
Notakikwaywin Lake	n.s.		n.s.		0		0	0.00
TOTAL & AVG. DENSITY**	8	0.04^	77	0.46^	0	0	83	0.46

*birds/km

** Excludes Notakikwaywin Lake

^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)

6.2 Hunter Interviews

Members of the Nelson House Band have traditionally hunted waterfowl during both spring and fall migrations. Waterfowl hunters expressed their concerns abut the effects of diversion during interviews conducted as part of this study (Figure 9). Hunters felt that the availability of ducks and geese at traditional hunting sites is considerably lower than it was before CRD. Hunters said that, before diversion, ducks and geese were plentiful throughout the area and hunters did not have to travel far to reach good hunting areas. They said they used to encounter numerous ducks along Rat River between Wapisu and Threepoint lakes and along the Burntwood River both above and below Threepoint Lake, and that upper reaches of Burntwood River in particular used to be frequented by young ducks. Ducks could usually be found in good numbers in shallower bays off Wapisu Lake, in the tall grass area that used to be located where Rat River entered Threepoint Lake, and in various locations in Threepoint Lake. Now, duck numbers are so low that hunters are forced to travel outside the Rat-Burntwood system to hunt waterfowl.

Hunters described at length how the increased water levels caused old shorelines to disappear and stands of trees to be inundated. Flooding of former shorelines has left fallen trees, logs and other debris in wide swaths along lake shores through much of the area. This has made boat navigation and shoreline access very difficult or impossible. Access problems have contributed to the reduction in opportunities to hunt waterfowl and moose. Hunters related how they would hear waterfowl or moose somewhere along the shoreline but, because of the debris, they could not move close enough to see or shoot animals. At times, even when it was possible to see ducks rise from dead trees standing along the shoreline, shooting the birds was pointless because birds often fell in areas where they could not be retrieved.

Extremely low numbers of waterfowl, combined with the difficulties of boat travel, have forced hunters to abandon their traditional hunting locations in the Rat-Burntwood area. Alternate hunting areas away from the diversion route are more difficult to get to and necessitate travelling greater distances resulting in increased travel costs.

WATERFOWL-RELATED CONCERNS AND ADVERSE EFFECTS

Observations of Nelson House Band Hunters*

Habitat

- used to be lots of grassy shallow-water areas in Threepoint Lake
- Honeymoon Creek used to be a narrow windy channel; now it's a lake
- old shorelines are gone; water is now into the trees

Waterfowl

- used to find young ducks along Burntwood River above Threepoint Lake
- used to see lots of ducks along Burntwood River downstream of Threepoint Lake, but not any more
- ducks used to be plentiful all along Rat River, but not any more
- bay at south end of Wapisu Lake was good for ducks and geese
- ducks and geese used to land in Threepoint Lake
- used to be lots of ducks in Wuskwatim Lake especially at south end

Hunting

- didn't have to travel far to reach good hunting areas
- debris in the water makes traveling by boat difficult
- traveling by boat at night is now too hazardous
- junk and debris along the shoreline makes ducks hard to see; when you do see them, you can't move close enough to them
- ducks that are shot cannot be retrieved due to trees and debris
- Threepoint Lake and Honeymoon Creek used to be good hunting areas
- used to be able to go to Honeymoon Lake and return with 10 ducks; now it's hard to see them let alone harvest them
- hunting along Rat River used to be good, but not any more
- ducks used to be plentiful in the back bays of Wapisu Lake
- creek entering northeast end of Footprint Lake used to be good; now you can't travel along it because of fallen trees
- used to be able to harvest ducks and geese on Threepoint Lake using blinds
- inland ponds are hard to get to; travel costs are greater

*Comments listed here represent the opinions of individual hunters

Figure 9. Waterfowl-related concerns and adverse effects identified during hunter interviews at Nelson House.

7.0 DISCUSSION

7.1 Pre- and Post-Diversion Waterfowl Use

Aerial surveys of shorelines in the Rat-Burntwood area were carried out before the Churchill River Diversion on September 7 and 26 in 1973 by Webb and Foster (1974) as part of the Lake Winnipeg, Churchill and Nelson Rivers Study. The September 7 survey was flown in a Cessna 337; a Cessna 180 was used for the September 26 survey. Survey altitudes ranged from 30 to 100 m and all bird counts were made within 150 m of shorelines. Honeymoon Lake, Notakikwaywin Lake and the reach of Burntwood River upstream of Threepoint Lake were not surveyed by Webb and Foster. Other water areas surveyed in 1973, in general, are comparable with those surveyed in 1988. Results of these surveys are summarized in Tables 6 through 9.

Comparison of 1973 and 1988 survey results for dabblers, divers, and all ducks combined (Tables 2-4 and 6-8) suggests that duck use along the Rat-Burntwood rivers system was considerably greater before diversion than after. Although more Canada geese were observed in 1973 than in 1988 (Table 9), their overall low numbers and spotty distribution make it difficult to draw any meaningful conclusions about the effects of CRD on fall staging populations.

Table 10 and Figures 10-14 show the percentage change in numbers of ducks that occurred between 1973 (pre-diversion) and 1988 (post-diversion) in the Rat-Burntwood area. The biggest changes in dabbler numbers occurred in Threepoint Lake (100 percent reduction) and Wuskwatim Lake (99.89 percent reduction) based on highest survey counts. The reduction in dabbler presence at the other three areas ranged from 85 to 95 percent. Numbers of divers were also lower in 1988 at Threepoint Lake (-96.8 percent), Wuskwatim Lake (-92.4 percent), Wapisu Lake (-90.9 percent), and lower Burntwood River (-80 percent); on Rat River, divers showed an increase (40 percent) in 1988, although counts there in both survey years were low.

Comparison survey results for dabblers, divers, and total ducks shows a substantial reduction in numbers from pre- to post-diversion periods over the Rat-Burntwood

Table 6. Numbers of dabbling ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	414	46	414
Rat River	200	87	200
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	15	377	377
Burntwood River (downstream)	57	502	502
Wuskwatim Lake	72	944	944
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	**758**	**1956**	**1956**

n.s. = not surveyed

Table 7. Numbers of diving ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	12	22	22
Rat River	10	0	10
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	20	31	31
Burntwood River (downstream)	5	0	5
Wuskwatim Lake	265	169	265
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	**312**	**222**	**312**

n.s. = not surveyed

Table 8. Numbers of ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	426	95	426
Rat River	235	87	235
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	35	408	408
Burntwood River (downstream)	97	502	502
Wuskwatim Lake	388	1127	1127
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	1181	2219	2219

n.s. = not surveyed

Table 9. Numbers of Canada geese observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	0	0	0
Rat River	90	20	90
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	6	90	90
Burntwood River (downstream)	0	0	0
Wuskwatim Lake	12	0	12
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	108	110	110

n.s. = not surveyed

Table 10. Changes in waterfowl numbers in the Rat-Burntwood rivers area from pre-diversion to post-diversion based on highest survey counts.

Species by Area	Pre-diversion 1973	Post-diversion 1988	% Change
WAPISU LAKE			
Canada Geese	0	0	0.0
Dabblers	414	25	-94.0
Divers	22	2	-90.9
All ducks*	426	27	-93.7
RAT RIVER			
Canada Geese	90	8	-91.1
Dabblers	200	30	-85.0
Divers	10	14	40.0
All ducks*	235	44	-81.3
THREEPOINT LAKE			
Canada Geese	90	0	-100.0
Dabblers	377	0	-100.0
Divers	31	1	-96.8
All ducks*	408	5	-98.8
LOWER BURNTWOOD R.			
Canada Geese	0	0	0.0
Dabblers	502	25	-95.0
Divers	5	1	-80.0
All ducks*	502	26	-94.8
WUSKWATIM LAKE			
Canada Geese	12	75	525.0
Dabblers	944	1	-99.9
Divers	265	20	-92.5
All ducks*	1127	20	-98.2

* includes dabblers, divers, & unidentified ducks.

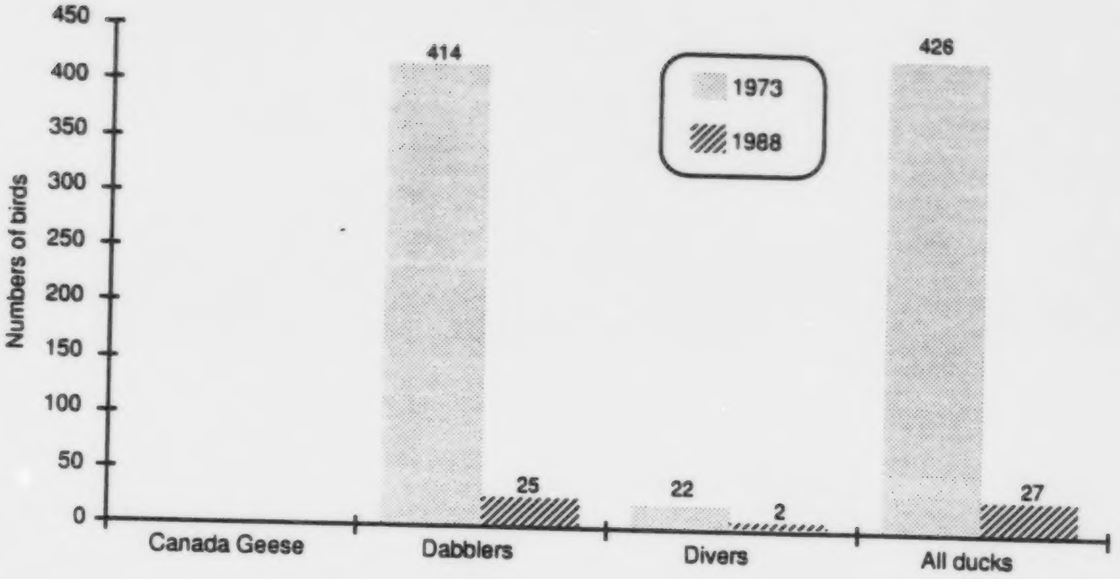

Figure 10. Pre- and post-diversion use of Wapisu Lake area by waterfowl based on highest survey counts.

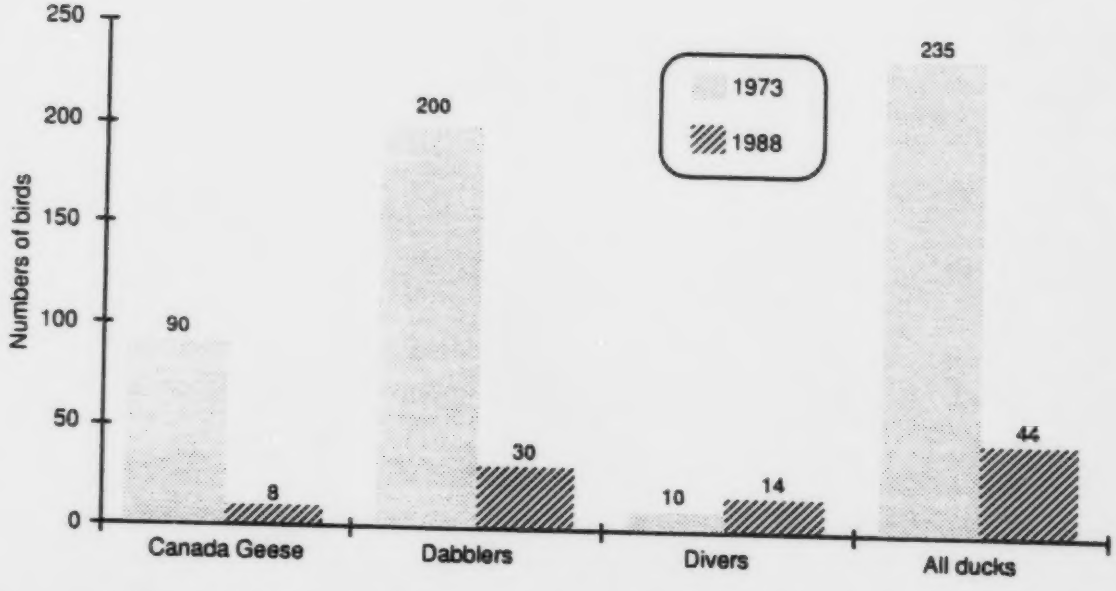

Figure 11. Pre- and post-diversion use of Rat River area by waterfowl based on highest survey counts.

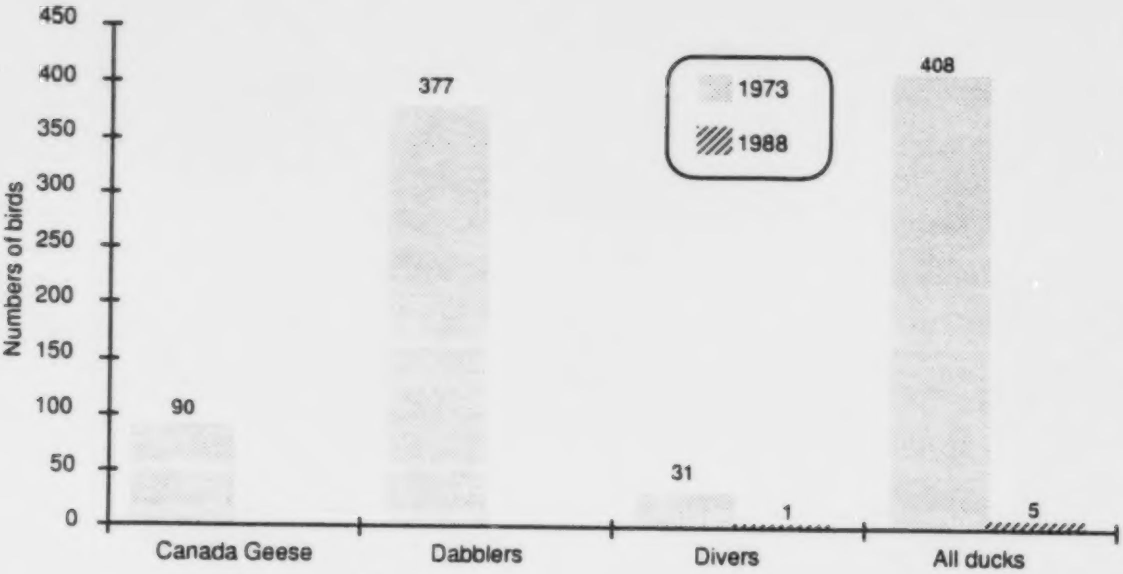

Figure 12. Pre- and post-diversion use of Threepoint Lake area by waterfowl based on highest survey counts.

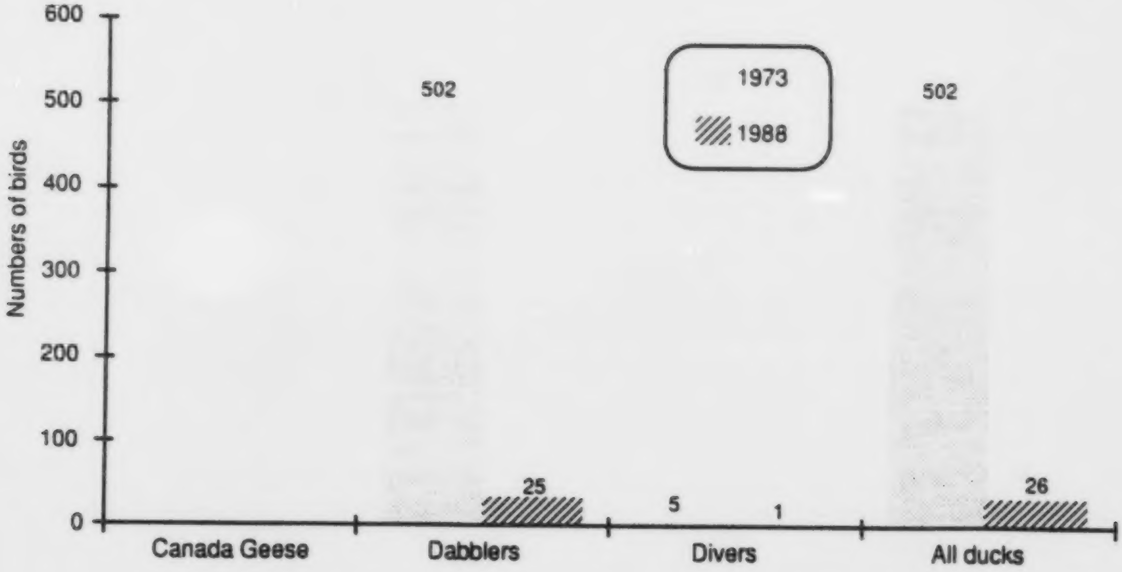

Figure 13. Pre- and post-diversion use of lower Burntwood River area by waterfowl based on highest survey counts.

system as a whole (Table 11, Figure 15), with dabblers accounting for the greatest change (-96.7 percent).

While numbers of Canada geese were 57 percent lower overall in the Rat-Burntwood areas in 1988, based on comparison of highest survey counts, one cannot reasonably assume that such change represents an actual decline in goose use, given the variability in observations between years and sites.

In general, it is risky to draw broad conclusions about changes in animal numbers from two years of survey data represented by a limited number of "point-in-time" counts. Nevertheless, in the case of dabbling ducks, the between-year differences are large enough to suggest use of the Rat-Burntwood system by dabblers has declined following diversion. The survey data also suggest that diving duck use of shoreline has declined since diversion, particularly in Wuskawatim Lake; however, as with Canada geese, the small numbers of divers recorded in both survey years make it difficult to state definitively that a decline has occurred in all survey areas.

7.2 Possible Causes of Waterfowl Use Changes

The following possible causes for the observed changes in waterfowl numbers, and especially of dabbling ducks, in the Rat-Burntwood system include:

1) Regional changes in waterfowl populations
2) Modifications to the physical environment and habitat associated with the Churchill River Diversion.

Table 11. Summary of changes in waterfowl use of the Rat-Burntwood area for lakes represented by both pre- and post-diversion counts.

Species	Pre-diversion 1973	Post-diversion^ 1988	% Change
Canada Geese	192	83	-57
Dabblers	2437	81	-97
Divers	333	38	-89
All ducks*	2698	122	-95

* includes dabblers, divers & unidentified ducks.

^ numbers of each species represent highest counts from Tables 2 through 5 and reflected in the post-diversion column of Table 10.

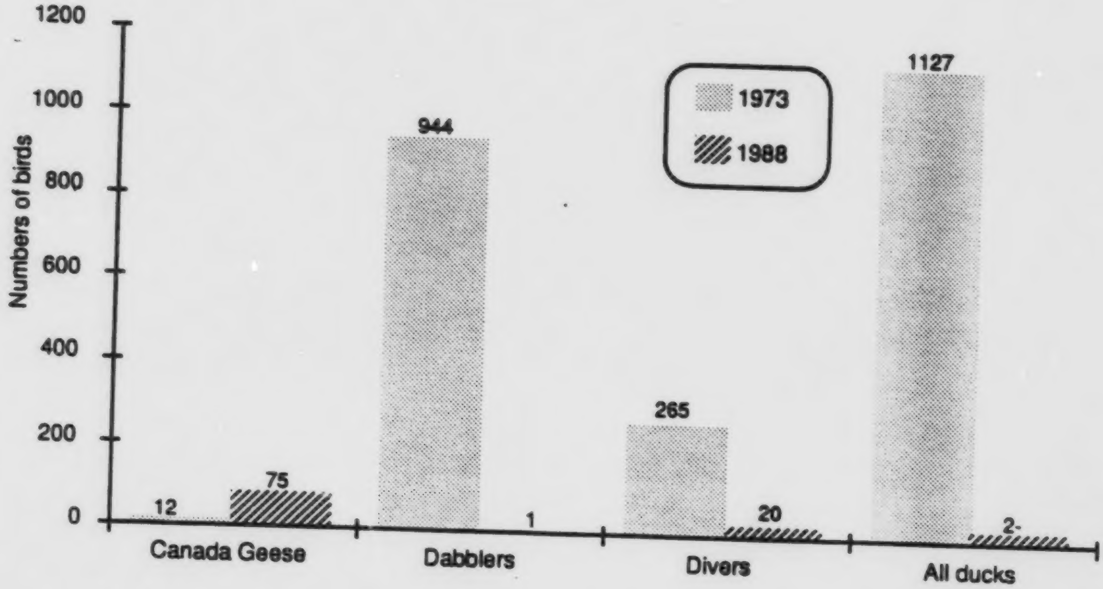

Figure 14. Pre- and post-diversion use of Wuskwatim Lake area by waterfowl based on highest survey counts.

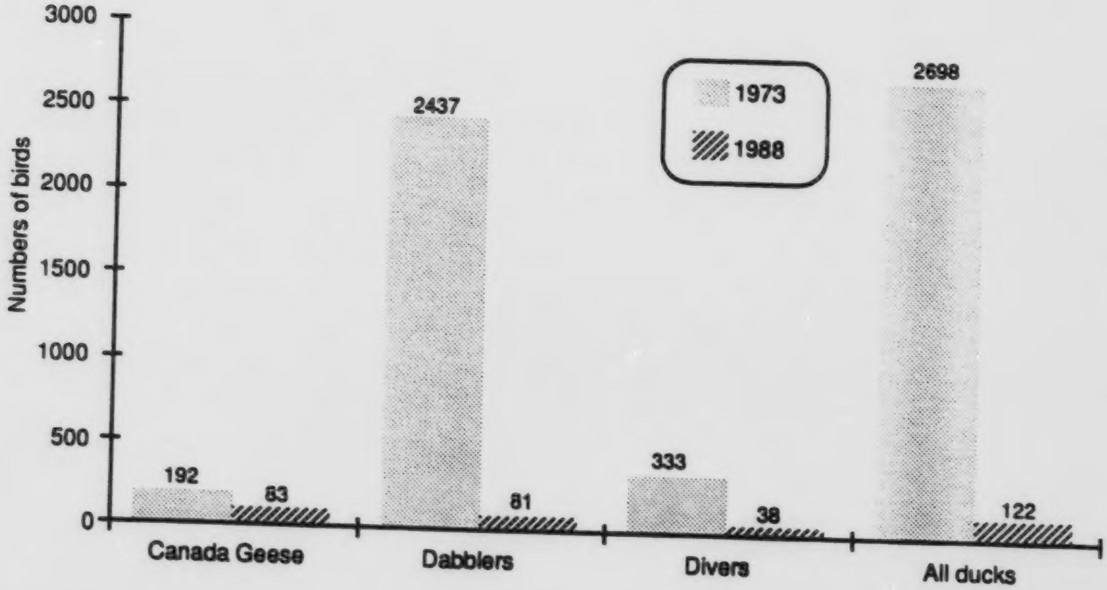

Figure 15. Pre- and post-diversion use of Rat-Burntwood area by waterfowl based on highest survey counts.

7.2.1 Regional Waterfowl Population Changes

The higher density of ducks recorded at Notakikwaywin Lake, located about 15 km from the Burntwood River system, indicates that the decline in duck use of the Rat-Burntwood rivers system following diversion is not simply a reflection of regional changes in duck populations. Nevertheless, one cannot overlook the possibility that regional declines in waterfowl populations could have occurred because of overexploitation by hunters, or as a result of a widespread reduction in the availability or quality of habitat. If waterfowl populations had declined throughout northern Manitoba, one would expect such declines to be represented in waterfowl numbers in more localized areas such as the Rat-Burntwood area.

To determine whether the observed reduction in waterfowl use of the Rat-Burntwood area could be attributed to a more widespread, regional decline in waterfowl populations, data collected annually by the U.S. Fish and Wildlife Service, during breeding pair surveys in Stratum 24, northern Manitoba (Figure 16), were examined. These data yield adjusted population estimates for ducks and indicated breeding pair numbers for Canada geese for the stratum. It must be stressed that these data estimate the relative size of breeding populations of ducks and geese and cannot be used to represent numbers or distribution of migrating or staging waterfowl. For the purposes of this study, the data are useful as broad indicators of regional population size and trends.

Data for the period 1964-75 were selected to describe regional population levels prior to the CRD project, while data for 1977-88 represent regional population levels following diversion. The diversion was phased in over 1976 and 1977 (J. Funnell, pers. comm.); data for those years were not included in the analysis. Duck and Canada goose population estimates for these two periods are given in Table 12. Regional population changes in ducks and Canada geese that have occurred from 1955 to 1988 are displayed in Figures 17 and 18.

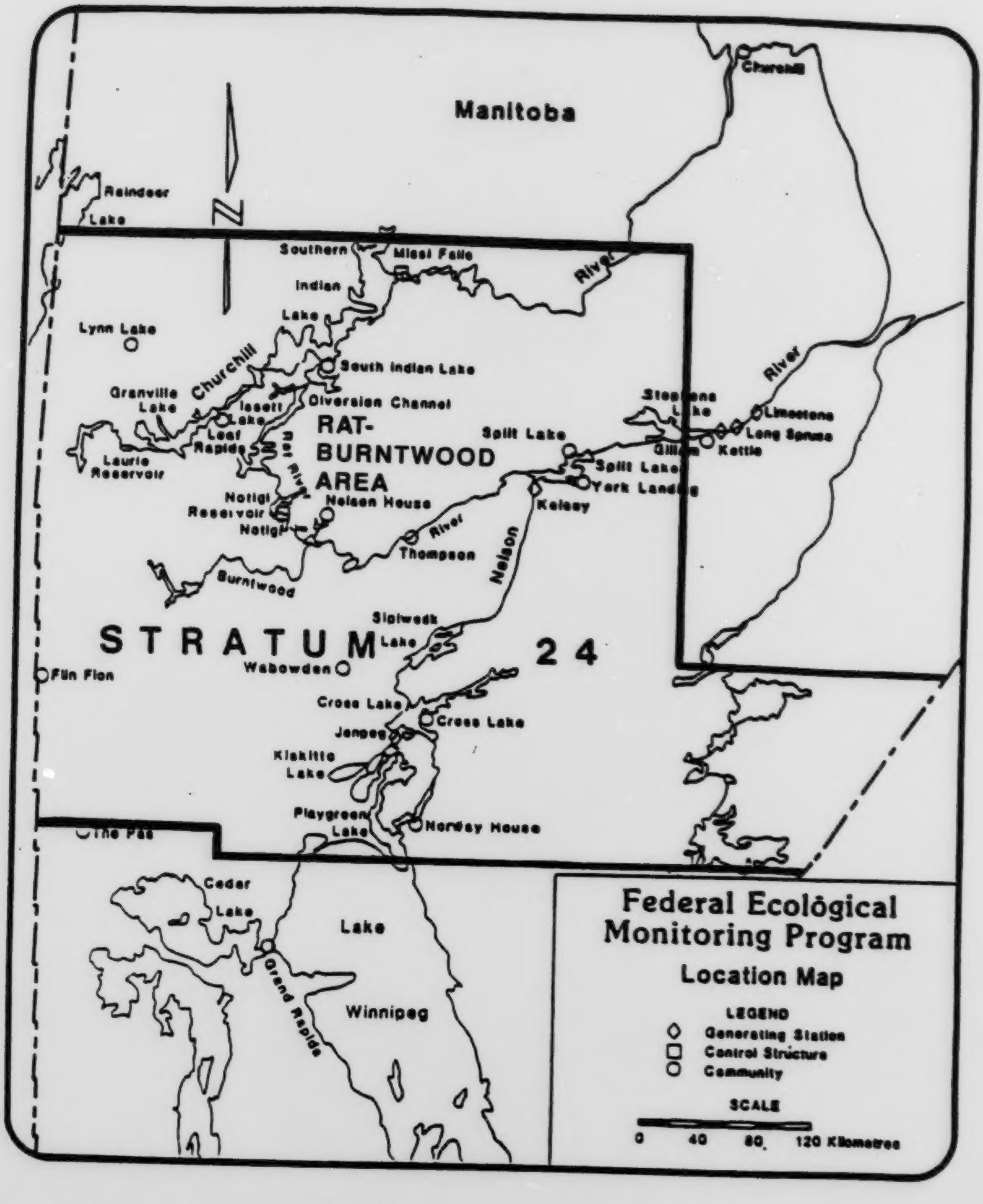
Manitoba
Churchill
N
Reindeer Lake
Southern Indian Lake
Missi Falls
River
Lynn Lake
Churchill
South Indian Lake
Granville Lake
Issett Lake
Diversion Channel
Leaf Rapids
Laurie Reservoir
Rat River
RAT-BURNTWOOD AREA
Notigi Reservoir
Notigi
Nelson House
Split Lake
Split Lake
York Land
Kelsey
Gill
Kettle
Long Spruce
Limestone
River
River
Thompson
Nelson
Burntwood
Sipiwesk Lake
STRATUM 24
Wabowden
Flin Flon
Cross Lake
Cross Lake
Jenpeg
Kiskitto Lake
Playgreen Lake
Norway House
The Pas
Cedar Lake
Grand Rapids
Lake Winnipeg
Federal Ecològical Monitoring Program
Location Map
LEGEND
◇ Generating Station
□ Control Structure
○ Community
SCALE
0 40 80 120 Kilometres

Figure 16. Location of waterfowl survey stratum 24.

Table 12. Adjusted population estimates (thousands) for dabblers, divers, and Canada geese in stratum 24 during pre-diversion (1964-75) and post-diversion (1977-88) periods (from U.S. Fish & Wildlife Service).

PRE-DIVERSION	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	MEAN
dabblers	228.9	404.4	205.0	716.0	1122.9	1102.3	1458.7	631.2	440.2	500.3	345.1	358.5	626.1
divers	198.2	276.7	228.3	233.0	587.7	653.3	788.0	447.4	513.8	375.5	270.2	383.2	412.9
Totals	**427.1**	**681.1**	**433.3**	**949.0**	**1710.6**	**1755.6**	**2246.7**	**1078.6**	**954.0**	**875.8**	**615.3**	**741.7**	**1039.1**
Canada geese	4560	6952	5145	5050	9504	15606	5925	4700	2498	1845	1590	2041	5451.3

POST-DIVERSION	1977	1978	1979	1980	1981	1982	1983	1983	1985	1986	1987	1988	MEAN
dabblers	708.9	1101.5	935.5	799.8	837.2	561.6	584.4	347.2	372.6	248.6	441.3	588.6	627.3
divers	536.2	376.2	336.9	387.6	746.3	329.4	531.7	319.4	359.1	534.1	339.8	288.2	423.7
Totals	**1245.1**	**1477.7**	**1272.4**	**1187.4**	**1583.5**	**891.0**	**1116.1**	**666.6**	**731.7**	**782.7**	**781.1**	**876.8**	**1051.0**
Canada geese	4444	6750	9743	5125	5681	2840	5000	2525	5517	1914	5345	8104	5249.0

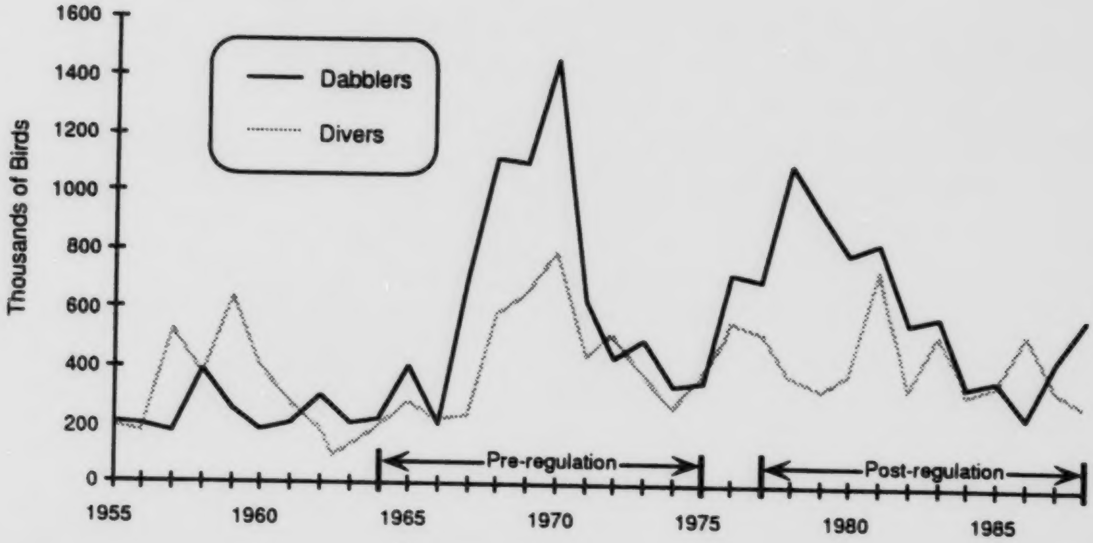

Figure 17. Adjusted population estimates of dabblers and divers in stratum 24 (northern Manitoba), 1955-1988.

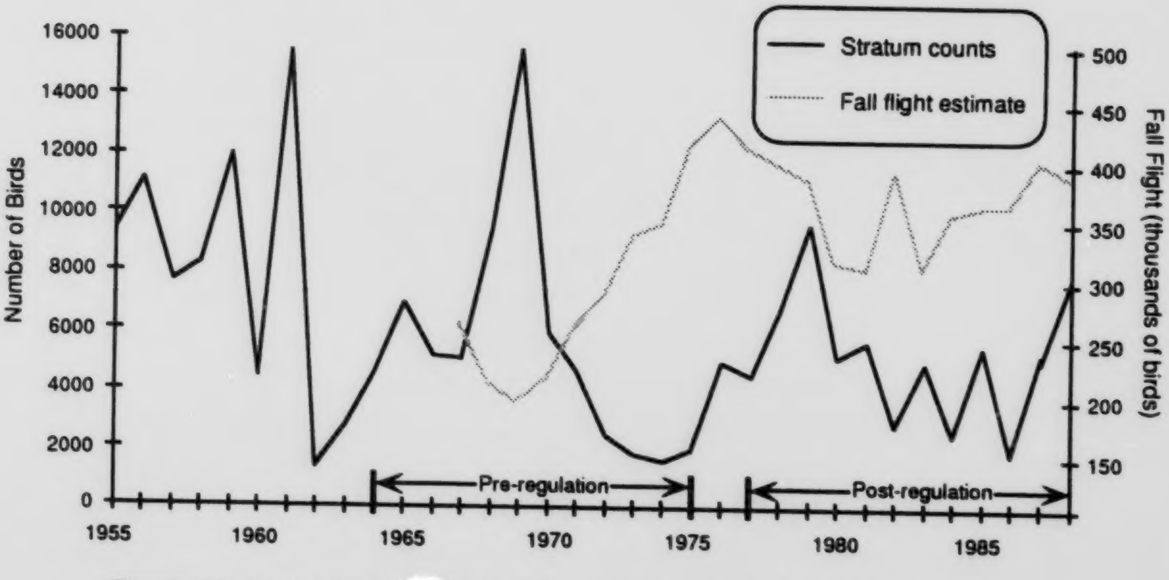

Figure 18. Estimates of indicated Canada goose pairs in stratum 24 (northern Manitoba), 1955-1988, and the estimated fall flight of Canada geese, Eastern Prairie Population, 1967-1988, based on production survey data.

Figure 17 shows that large year-to-year fluctuations in regional dabbler and diver populations occurred during both the pre- and post-diversion periods. However, when the 12-year means for the two periods are compared, only a small change in numbers is evident from the pre-diversion period to the post-diversion period for both dabblers and divers (Table 13). Results of the 1973 and 1988 surveys for the Burntwood area included for comparison. Overall, the dabbler population increased by 0.2 percent across Stratum 24 while the diver population declined by 2.6 percent.

One must keep in mind, however, that the means on which these changes are based will vary somewhat as individual year's counts are added or subtracted. The main point to be made is that numbers of breeding waterfowl in the study region have neither declined nor increased over the long term, according to Stratum counts.

Figure 18 shows that, as with regional dabbler and diver populations, large year-to-year fluctuations in the Canada goose breeding population have occurred in stratum 24 during both the pre-diversion and post-diversion periods, but do not indicate any long-term trends toward population increase or decline.

For comparison, fall-flight estimates for the Eastern Prairie Population (EPP) of Canada Geese based on production surveys carried over an area that broadly overlaps Stratum 24 (EPP Progress Report, 1988) have also been included in Figure 18. Disregarding the obvious differences between the two sets of data, there is no indication of a downward trend in the EPP fall-flight forecasts, which have remained relatively stable -- with moderate between-year fluctuations -- since 1975.

In conclusion, there is no indication that the observed reduction in waterfowl use in the Rat-Burntwood area reflects a long-term decline in regional populations as represented by the Stratum 24 counts, since no such trends are evident in the stratum data. At the same time, one must be cautious about interpreting the significance of the 97 percent change in dabbler numbers derived from 1973 and 1988 surveys, because large between-year changes are not uncommon in waterfowl counts derived from aerial surveys.

Table 13. Changes in regional (Stratum 24) and local (Rat-Burntwood) waterfowl use from pre-diversion (1973) to post-diversion (1988).

Waterfowl	Stratum 24: 12-year Means* 1965-75 (pre)	1978-88 (post)	% Change	Rat-Burntwood Counts** 1973 (pre)	1988 (post)	% Change
Dabblers	626.1	627.3	0.2	2437	81	-96.7
Divers	412.9	423.7	2.6	333	38	-88.6

* adjusted population estimates (thousands)
** numbers represent birds/km

For example, comparison of any pre-diversion year with any post-diversion year estimate using the Stratum 24 data can yield dramatically different results that may or may not have anything to do with changes in regional water conditions: comparison 1986 and 1970 Stratum counts suggest a 78 percent reduction in dabbling ducks, whereas comparison of 1978 and 1966 counts indicate a 437 percent increase. In the absence of counts compiled over several years, one would be hard-pressed to draw meaningful conclusions from these point-in-time data.

7.2.2 Modifications to the Habitat

Extensive modifications to terrestrial and aquatic habitat have occurred due to increases in water levels that have accompanied the diversion. The magnitude of these increases is illustrated by the water level data collected at Water Survey of Canada's Footprint Lake at Nelson House hydrometric station (# 05TF001). The historic record from 1969 to 1988 was divided into pre-diversion (1969-75) and post diversion (1978-88) periods. Data for 1976 and 1977 were excluded because they were considered to be transitional between pre- and post-diversion conditions. Connection of Southern Indian Lake to Rat River occurred on June 2, 1976, and the diversion was phased in over 1976 and 1977 (J. Funnell, pers. comm.).

Mean monthly water levels at Footprint Lake during the pre- and post-diversion periods are presented in Table 14 and Figure 19. These data show that, when the historic record is considered, mean monthly post-diversion water levels were between 3.81 and 5.45 m higher than mean monthly pre-diversion water levels. In August and September, when waterfowl have traditionally used the Rat-Burntwood area for staging prior to migration, mean monthly post-diversion water levels were approximately 4.5 m higher than the mean levels before the diversion. (Because gaps exist in the daily record of water levels at Footprint Lake, the above means must be considered approximate).

These comparisons indicate that, following diversion, water levels along the diversion route rose considerably. The effect of this increase in water levels on the environment has been experienced by members of the Nelson House Band and was described during the hunter interviews (see section 6.2 of this report). The changes to shoreline habitat resulting from the rise in water levels were also noted during the aerial waterfowl surveys. Well-vegetated marsh areas that used to exist in shallow bays of lakes, such as Wapisu Lake, and along the Rat and Burntwood rivers, have been inundated. In contrast, abundant marsh habitat was observed in shallow water areas of Notakikwaywin Lake which remains unaffected by the diversion. Pondweeds and other submerged vegetation were clearly visible at Notakikwaywin Lake but were not observed anywhere in the Rat-Burntwood area.

Table 14. Pre- and post-diversion water levels (m asl) at Footprint Lake-Nelson House hydrometric station (#05TF001).

	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
A. HISTORIC												
1978-88 (Post)	243.42	243.26	243.08	242.89	242.81	242.98	243.05	243.18	243.05	243.12	243.22	243.27
1969-75 (Pre)	238.02	237.82	237.63	237.70	238.89	239.17	239.07	238.65	238.51	238.66	238.57	238.45
Difference	5.40	5.44	5.45	5.19	3.92	3.81	3.98	4.53	4.54	4.46	4.65	4.82

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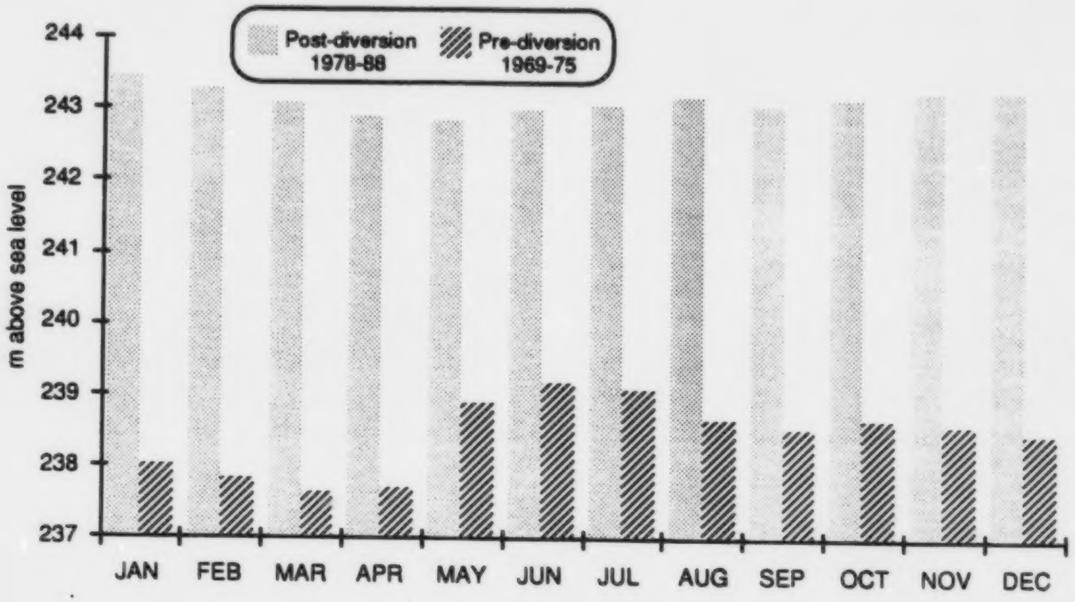
* Due to gaps in monthly water level records, monthly means could not be calculated for every year; therefore the multi-year means shown here were derived from discontinuous data and must be considered approximate.

BASED ON HISTORIC RECORD



BASED ON SELECTED YEARS

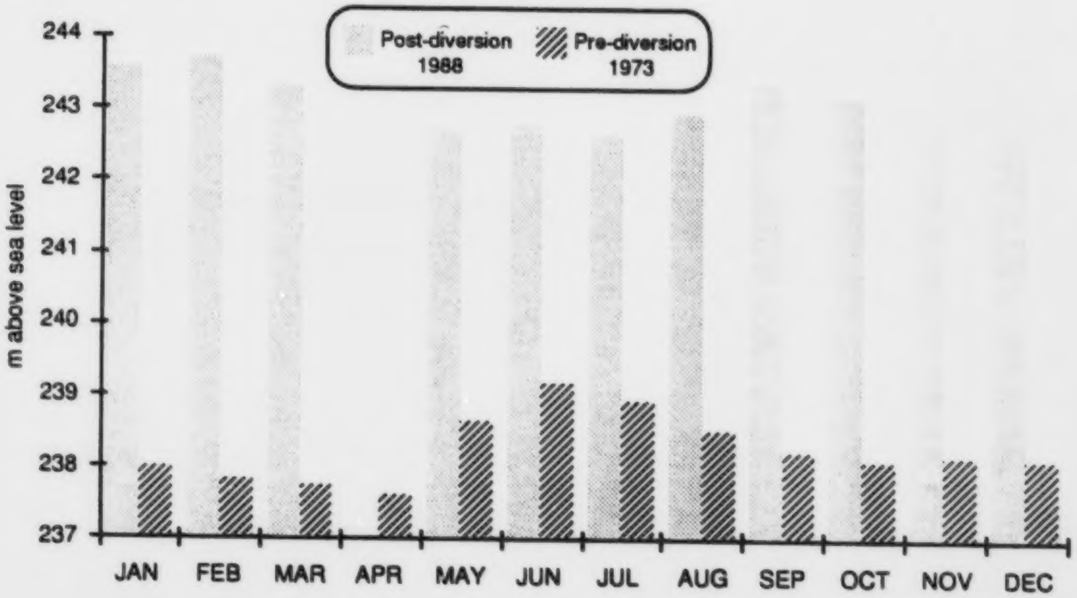


Figure 19. Pre- and post diversion water levels at Footprint Lake (Nelson House) hydrometric station (#05TF001).

While former shallow water areas have been transformed into deep water areas, a large proportion of new shorelines associated with shallow water are ill-defined and are characterized by dead trees (fallen or standing), tree stumps and other debris -- habitat not conducive to attracting waterfowl.

In their assessment of shoreline conditions on Footprint and Honeymoon lakes, Kellerhals Engineering (1988) noted that before diversion, areas with shallow water depths or floating or submerged vegetation were limited to low-gradient deltas near mouths of streams, with cleared or eroded shorelines confined to the Nelson House area. After diversion, the length of shoreline characterized by shallow water or floating/submerged vegetation increased from 14 km (pre-diversion) to 83 km. The shoreline length with standing debris rose from 0.2 to 56 km.

From their analysis, Kellerhals Engineering (1988) concluded:

> *that the Churchill-Nelson Diversion has had a widespread impact on shoreline characteristics. The extensive areas of newly eroding shoreline will act as a continuing source of sediment until such time as they restabilize. Areas of shallow water, floating or submerged vegetation and inundated debris restrict access to the shore zone and dead-heads will interfere with open-water travel for years to come.*

It is important to remember, however, that, in the long term, increases in the shallow-water zone associated with higher water levels can be expected to provide at least as much habitat suitable for waterfowl (including diving ducks) as existed before regulation, although the time required for shorelines to stabilize will span several decades.

The key habitat attributes that formerly attracted ducks prior to fall migration are now absent. The dramatic reduction in use of the area by waterfowl would appear to support the idea that habitat appearance is a major factor influencing waterfowl use.

The influence of landscape and habitat characteristics on the attractiveness of an area to waterfowl and other bird species was discussed by I.D. Systems (1991) in relation to the effects of the Lake Winnipeg Regulation project on waterfowl use of the Outlet Lakes area. Some of this discussion is germane to understanding the effect of habitat modifications, associated with the CRD, on waterfowl use of the Rat-Burntwood area and is repeated here.

The mechanics of bird navigation and habitat selection is poorly understood despite the fact that many scientific papers have been published on these subjects (Bellrose 1976). Hilden (1965) provides some useful insights in his review of habitat selection mechanisms in birds during the breeding season. It is reasonable to assume that similar mechanisms operate with respect to selection of staging habitat by waterfowl during fall migration.

Hilden refers to "proximate" and "ultimate" factors that together determine the selection of suitable habitat. Ultimate factors such as food and shelter are essential to the survival of a species. Proximate factors refer to characteristic species-specific stimuli provided by habitat that release the "settling reaction". These factors include: landscape, terrain, feeding and drinking sites, and other animals.

Not every suitable habitat need possess all the features characteristic of the optimal environment. Rather, the positive and negative habitat characteristics are summed and, when the combined effect of the stimuli exceeds the threshold of the settling reaction, the habitat is selected. Similarly, one key stimulus, such as adverse landscape appearance, may out-weigh the other stimuli; in that situation, the combined effect of other positive stimuli would not be sufficient to induce the bird to "settle" in that particular habitat. Studies on wild geese conducted by Barry (1962) and Wood (1964) concluded that, where certain proximate factors such as appropriate nesting habitat conditions are lacking, ovulation can be inhibited and the nesting cycle terminated. Marshall (1952) speculated that inhibitions to breeding in birds appear to be brought about by the lack of one or more important environmental stimuli necessary to complete the reproductive cycle.

Habitat selection is described by Hilden as being a two-stage process. The first stage, consisting of settling down and exploring a particular habitat, is released by features of the landscape and general characteristics of the terrain. Whether the area in question is chosen or rejected depends on how closely certain of its details conform to other proximate factors in the bird's habitat selection mechanism.

The Rat-Burntwood area provides numerous environmental stimuli related to habitat quality and appearance that would likely cause ducks and geese to seek other more suitable areas. These negative stimuli include: the absence of submerged vegetation; the absence of emergent aquatic vegetation; the presence of inundated trees and debris along the shoreline; and the presence of eroding and slumping banks along the shoreline. The only positive environmental stimulus from the standpoint of waterfowl is the presence of water, which in the Rat-Burntwood area apparently is not sufficient to elicit a "settling reaction" in fall staging waterfowl.

8.0 SUMMARY AND CONCLUSIONS

8.1 Project Impacts on Waterfowl

Following completion of the Churchill River Diversion project in 1977, waterfowl use of the Rat-Burntwood rivers system in the fall has declined. Numbers of ducks observed along shorelines of lakes and rivers in the Rat-Burntwood area in the fall of 1988 were approximately 95 percent lower than the numbers recorded prior to the CRD project in the fall of 1973. Dabbling ducks have experienced the greatest reduction - almost 97 percent - while diving ducks declined by about 89 percent. Canada goose use of the area dropped by 57 percent, although this change is of questionable significance, given the limited number and variable distribution of birds recorded during 1973 and 1988 surveys.

The observed decline in the use of the Rat-Burntwood area by dabblers in particular cannot be attributed to regional waterfowl populations documented in northern Manitoba. There is no indication that regional duck populations in the study region have increased or declined in the long term, although their numbers have fluctuated dramatically from year to year.

Analysis of water level data collected at Footprint Lake indicates that, following the diversion, water levels along the Rat-Burntwood system have risen 3.5 to 5.8 m. The elevated water levels have flooded natural shorelines and created poor quality habitat characterized by dead trees, fallen timber and other debris. As a result, habitat modifications resulting from the Churchill River Diversion may have rendered the Rat-Burntwood less attractive for staging dabblers and (to a lesser extent) divers.

8.2 Project Impacts on the Nelson House Band

Nelson House residents interviewed during the study indicated that presence of fallen trees and other debris has made boat travel and access to traditional waterfowl harvesting areas difficult or impossible. The reduction in waterfowl availability, combined with boat navigation and access problems, have limited opportunities for Nelson House Band members to hunt and view waterfowl.

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APPENDIX A

NORTHERN FLOOD AGREEMENT

Background

The **Northern Flood Agreement** (NFA) was signed on December 16, 1977 by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee to establish a mechanism for compensating members of the Nelson House, Norway House, Cross Lake, Split Lake and York Landing (formerly York Factory) bands for adverse effects caused by the hydroelectric development.

The Preamble of the Agreement lays out the premises and principles which together provide a basis for the Agreement's provisions. Clauses C, D, E and G of the Preamble are particularly relevant to the present study and are reproduced below verbatim:

> *C.* *As a result of the modification of the water regime, adverse effects have occurred and may continue to occur, on the lands, pursuits, activities and lifestyles, of the residents, individually and collectively, of the Reserves of Cross Lake, Nelson House, Norway House, Split Lake and York Landing ("the Reserves" as hereinafter defined);*
>
> *D.* *The parties wish to ensure that all persons as defined herein, who may be, or have been, directly or indirectly, adversely affected by the* [Lake Winnipeg Regulation and Churchill River Diversion] *Project shall be dealt with fairly and equitably;*
>
> *E.* *Uncertainty as to the effects of the Project, with respect not only to the Project as it exists at the date of this Agreement but also as it may develop in he future, is such that it is not possible to foresee all the adverse results of the Project nor to determine all those persons who may be affected by it, and, therefore it is desirable to establish through the offices of a single arbitrator a continuing arbitration instrument, to which any person may submit a claim,*

and as well as to fully empower such arbitrator to fashion a just and appropriate remedy;

G. *Canada, by virtue of its jurisdiction and responsibility for Indians and land reserved for Indians, is committed to playing an active role in providing opportunity for the continued viability of the communities and, in particular but without limitation, in making available resources and expertise to the communities in planning and improving the social and economic conditions of the communities, and in ensuring that the special rights of Indians, including those arising from Treaty 5, are adequately protected.*

Contractual Obligations of the Northern Flood Agreement

The provisions of the NFA are arranged in a set of 23 Articles which represent contractual obligations. Three subarticles ultimately lead to the initiation of the present study and are, therefore, reproduced below verbatim:

17.1 *Hydro and Canada and Manitoba, severally and jointly, undertake to implement such recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board Report which affect the communities and which fall within their respective or joint jurisdictions.*

17.5 *In particular but without limitation, monitoring of adverse effects of the Project pursuant to the Lake Winnipeg, Churchill and Nelson Rivers Study Board recommendations shall be planned and implemented so as to provide information as may be necessary to give effect to this Agreement.*

18.2 *Canada and Manitoba recognize that the Project is intended to benefit all citizens of Canada, and most particularly of Manitoba, on the one hand, and that the resource users have been and may continue to be adversely affected on the other hand, and that it is in the public interest to ensure that any damage to the interests, opportunities, lifestyles and assets of those adversely affected be compensated appropriately and justly.*

Claim 18

In December 1981, the five NFA Indian Bands and the Northern Flood Committee filed a claim (Claim 18) alleging that Canada, Manitoba and Manitoba Hydro (the Respondents) had failed to implement recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board and to report to the NFA bands on implementation measures.

In particular, they claimed that the Respondents had "failed to establish a mechanism to deal with monitoring and analysis of ongoing social and economic changes related to hydro-electric development and more generally, northern development pursuant to Recommendation 5(c) of the Study Board". In addition, they claimed that the Respondents had "failed to develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the area affected by the Hydro Project pursuant to Recommendation 10 of the Study Board".

Failure to implement these two recommendations meant that the adverse ecological effects of the Project, and the associated adverse social and economic effects, had not been monitored and the information necessary to give effect to the NFA not provided. Consequently, the requirements of NFA sub-article 17.5 had not been met. Furthermore, without this information, it has not been possible to determine to what extent damages to the interests, opportunities, lifestyles and assets of resource users have occurred and, therefore, the level of compensation that is necessary (sub-article 18.2).

Filing of the claim resulted ultimately in approval of funds by Treasury Board on February 6, 1986, for implementation of the NFA over the period 1986-91 (T.B. Minute 800953). Availability of these funds permitted the Canadian Wildlife Service to initiate studies of the adverse effects of the project on waterfowl use of the Rat-Burntwood rivers system in the fall of 1988.

Impact Management

Impact management refers to a mechanism, or set of mechanisms, by which the adverse impacts of the CRD project may be managed, whether they be biophysical impacts (e.g., adverse effects on waterfowl) or socio-economic-cultural impacts (e.g., reduced waterfowl hunting opportunities). Suitable and adequate compensation for adverse effects on NFA band would constitute redress for damages suffered by the bands.

NFA sub-article 24.8 refers to three forms of compensation or redress for damages suffered:

> 24.8 *Because mitigatory and/or remedial measures are more likely to have a lasting beneficial effect on the viability of a community and/or on individual residents than monetary compensation, such measures shall be preferred and only where mitigatory and/or remedial measures are not feasible or fail in effectiveness shall monetary compensation be ordered in lieu thereof in respect of any adverse effect.*

Article 1 of the NFA provides definitions for the terms "mitigatory measure" and "remedial measure":

> 1.9 *"Mitigatory measure" means any work, program or measure which is designed or intended to diminish, prevent, or ameliorate any adverse effect of the Project.*
>
> 1.14 *"Remedial measure" means any work, program or measure which is designed or intended to enhance, preserve, restore or replace in kind, wholly or in part, any property, land, land use interest or activity of any person, which has been or may be adversely affected by the Project.*

The third form of compensation is "monetary settlement" and would consist of a cash settlement for damages suffered.

NFA sub-article 24.6 provides the Arbitrator with "broad authority and power to make [compensation] awards capable of implementation and to fashion an appropriate and just remedy of any and all adverse effects of the Project on any person". Compensation awards are made on the basis of claims submitted to the Arbitrator for settlement. It should be noted that Claim 18 was not advanced for the purpose of obtaining a suitable remedy for the adverse effects experience. Rather, it was advanced to give effect to sub-article 17.5 (information) and sub-article 18.2 (compensation). The purpose of the claim was to cause the necessary information to be collected and made available which would allow adverse effects on NFA bands, and therefore damages, to be determined. This information would then be used to prepare and advance separate claims for compensation of damages.

EFFECTS OF THE LAKE WINNIPEG REGULATION PROJECT ON WATERFOWL USE OF THE OUTLET LAKES AREA AND ON NORWAY HOUSE BAND

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ABSTRACT

Surveys carried out in September 1986 and October 1987 indicated that use of the Outlet Lakes area by diving ducks (Aythyinae) in fall declined following completion of the Lake Winnipeg Regulation project. Diving duck numbers in 1986/87 were 96 percent lower than recorded during 1972/73 surveys.

The decline in diving duck use of the Outlet Lakes area cannot be attributed to a regional reduction in waterfowl populations, since no such decline was evident in regional survey data (stratum 24) compiled over the past thirty years. Analysis of water level data for Playgreen Lake indicates that slight year-round increases in water levels following Lake Winnipeg regulation have altered the quality of habitat and may have reduced the attractiveness of affected lakes to diving ducks in the fall.

Residents of Norway House interviewed during the study indicated that low diving duck numbers and high water levels have reduced opportunities for Band members to hunt and view waterfowl.

TABLE OF CONTENTS

Page

ABSTRACT

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION 1

2. STUDY AREA 3

3. LAKE WINNIPEG REGULATION PROJECT 5

4. PRE-PROJECT ENVIRONMENTAL ASSESSMENT 7

4.1 Lake Winnipeg, Churchill and Nelson Rivers Study 7

4.1.1 Study Findings 7
4.1.2 Study Board Recommendations 8

4.2 Canadian Wildlife Service Studies 10

4.2.1 Kiskittogisu River 10
4.2.2 Kiskitto Lake 10

4.3 Kiskitto Lake Regulation Committee Study 11

5. METHODS 12

5.1 Impact Assessment and Impact Management Approach 12

5.1.1 Biophysical Impacts 12
5.1.2 Socio-Cultural-Economic Impacts 14
5.1.3 Impact Management 15

TABLE OF CONTENTS (Cont'd)

Page

5.2 Data Collection 15

5.2.1 Effects on Waterfowl and Habitat 15
5.2.2 Effects on Norway House Band 18

6. RESULTS 19

6.1 General Habitat Conditions 19
6.2 Waterfowl Use 19
6.3 Hunter Interviews 24

7. DISCUSSION 30

7.1 Pre- and Post-Regulation Waterfowl Use 30
7.2 Possible Causes of Waterfowl Use Changes 39

7.2.1 Regional Waterfowl Population Changes 39
7.2.2 Transmission Lines and Channels 45
7.2.3 Habitat Changes 47

7.3 Assessment of Habitat Change Impacts on Waterfowl 50

7.3.1 Methodology 50
7.3.2 Valued Ecosystem Components 50
7.3.3 Impact Hypotheses 53

7.3.3.1 Impact Hypothesis A 54
7.3.3.2 Impact Hypothesis B 60

8. SUMMARY AND CONCLUSIONS 63

8.1 Project Impacts on Waterfowl and Waterfowl Habitat 63

8.2 Project Impacts on the Norway House Band 63

9. LITERATURE CITED 65

APPENDICES

LIST OF TABLES

Table — Page

1. Lengths (km) of post-regulation surveys of the Outlet Lakes area. 17
2. Numbers and densities of diving ducks observed during post-regulation surveys of the Outlet Lakes area. 23
3. Numbers and densities of dabbling ducks observed during post-regulation surveys of the Outlet Lakes area. 23
4. Numbers and densities of ducks observed during post-regulation surveys of the Outlet Lakes area. 27
5. Numbers and densities of Canada geese observed during post-regulation surveys of the Outlet Lakes area. 27
6. Lengths (km) of pre-regulation surveys of the Outlet Lakes area. 31
7. Numbers and densities of diving ducks observed during pre-regulation surveys of the Outlet Lakes area. 32
8. Numbers and densities of dabbling ducks observed during pre-regulation surveys of the Outlet Lakes area. 32
9. Numbers and densities of ducks observed during pre-regulation surveys of the Outlet Lakes area. 34
10. Numbers of densities of Canada geese observed during pre-regulation surveys of the Outlet Lakes area. 34
11. Changes in waterfowl densities in the Outlet Lakes area from pre-regulation to post-regulation based on highest survey densities. 38
12. Summary of changes in waterfowl use of the Outlet Lakes area from pre-regulation to post-regulation based on highest survey densities. 38
13. Adjusted population estimates (thousands) for dabblers, divers, and Canada geese in Statum 24 during pre-regulation (1964-75) and post-regulation (1977-88) periods (from U.S. Fish and Wildlife Service). 42
14. Changes in regional (Stratum 24) and local (Outlet Lakes) waterfowl use from pre-regulation to post-regulation. 44

TABLES (Cont'd)

Page

15. Pre- and post-regulation water levels (m asl) at Playgreen Lake - Norway House hydrometric Station (#05UB001). 49

16. Food items (by % of diet) consumed by lesser scaup in Alberta, Saskatchewan, Manitoba, Northwest Territories, Minnesota, and Illinois. 52

17. Numbers of insect species observed on broad-leaved and narrow-leaved pondweeds (Berg 1949). 58

LIST OF FIGURES

Figure		Page
1.	Outlet Lakes study area.	4
2.	The impact assessment and impact management approach used in the study.	13
3.	Waterfowl survey route, 1986/87.	16
4.	Location of the 8-Mile Channel in relation to Kiskittogisu River, Kiskittogisu Lake and Playgreen Lake (south basin).	20
5.	Diving and dabbling duck use of the Outlet Lakes area in Fall, 1986/1987.	22
6.	Use of the Outlet Lakes area by ducks (divers, dabblers, unidentified) and Canada geese in Fall, 1986/1987	25
7.	Waterfowl-related concerns and adverse effects identified during hunter interviews at Norway House.	26
8.	Traditional fall waterfowl hunting areas of the Norway House Band in the Outlet Lakes area.	29
9.	Pre- and post-regulation use of Playgreen Lake south basin by waterfowl based on highest survey densities.	36
10.	Pre- and post-regulation use of Playgreen Lake north basin by waterfowl based on highest survey densities.	36
11.	Pre- and post-regulation use of Kiskitto Lake by waterfowl based on highest survey densities.	37
12.	Pre- and post-regulation use of Kiskittogisu Lake by waterfowl based on highest survey densities.	37

FIGURES (Cont'd)

Page

13. Location of waterfowl survey stratum 24. 40

14. Adjusted population estimates of dabblers and divers in Stratum 24 (northern Manitoba), 1955-88. 43

15. Network diagram showing relationship between waterfowl use and various influencing environmental factors in the Playgreen Lake area. 56

1. INTRODUCTION

In the early 1970's, Manitoba Hydro began to proceed with developments to harness the hydroelectric potential of the Nelson River. This involved the diversion of a major portion of the lower Churchill River into the Nelson River, referred to as the Churchill River Diversion (CRD) Project, and regulation of the outflow from Lake Winnipeg through the Lake Winnipeg Regulation (LWR) Project. The LWR project is described in detail in **3. LAKE WINNIPEG REGULATION PROJECT.**

In response to concerns expressed about the effects of this development scheme on the natural environment and on northern communities that depended on the natural resources, the governments of Canada and Manitoba initiated the Lake Winnipeg, Churchill and Nelson Rivers Study. In April 1975, a report was issued by the Lake Winnipeg, Churchill and Nelson Rivers Study Board (Study Board or LWCNRSB) which described the environmental and social impacts that were expected to occur as a result of the hydroelectric development. The report also recommended measures to minimize or offset impacts and specified additional studies that should be carried out. Study Board findings and recommendations relevant to the present study are discussed in **4. PREVIOUS ENVIRONMENTAL ASSESSMENT STUDIES.**

To ensure that all members of Indian communities affected by the hydroelectric development would be adequately compensated, the **Northern Flood Agreement (NFA)** was entered into on December 16, 1977, by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee. The Northern Flood Committee was established to represent the interests of Nelson House, Norway House, Cross Lake, Split Lake and York Factory Indian Bands. Included in the NFA is the requirement that the effects of the development be monitored to determine how Indian Band members have been affected and how they should be compensated.

The NFA also established an arbitration procedure whereby injured parties could file claims and obtain suitable remedies for damages suffered. One such claim, Claim 18, led to the initiation of a Federal Ecological Monitoring Program (FEMP) in the spring of 1986. FEMP was established to address federal responsibilities under the NFA with respect to environmental research and monitoring. Claim 18 and the various provisions of the NFA are described further in **APPENDIX A: NORTHERN FLOOD AGREEMENT**.

Funds provided through FEMP enabled the present study to be initiated in response to concerns expressed by Norway House Band that opportunities for hunting waterfowl in the fall had declined following completion of the LWR project.

2. STUDY AREA

For the purposes of this study, the Outlet Lakes area consists of Playgreen, Kiskittogisu and Kiskitto lakes, bounded by Lake Winnipeg to the south and the Jenpeg Generating Station to the north (Figure 1). Cross Lake was excluded because it falls outside the study area. The Outlet Lakes receive water from the Red, Winnipeg, Saskatchewan and Dauphin river drainage basins via Lake Winnipeg.

The topography of the area reflects the underlying bedrock and the area's geomorphic history, particularly that of glacial and post-glacial periods (LWCN-RSB 1975a). The distribution of surface material has been influenced by glaciation and glacial Lake Agassiz. Organic deposits, glacial till, lacustrine clays and glaciofluvial sands comprise the surficial material. Organic deposits are widespread and consist mainly of peat. The three study lakes feature irregular bedrock shorelines that form promontories, and erodible organic and granular shorelines forming continuous sweeping arcs (LWCNRSB 1975b).

Black spruce (*Picea mariana)* and tamarack (*Larix laricina*) dominate most forest communities. Jack pine (*Pinus banksiana*), trembling aspen (*Populus tremuloides*), white spruce (*Picea glauca*), balsam poplar (*Populas balsamifera*), balsam fir (*Abies balsamea*), white birch (*Betula papyrifera*) and alder (*Alnus* spp.) occur less frequently. Willow (*Salix* spp.), alder and sedge (*Carex* spp.) communities inhabit wet or saturated sites (LWCNRSB 1975b). Marsh vegetation before regulation was characterized by a band of sedge along shorelines with an outer band of bulrush (*Scirpus* spp.) in deeper water. Floating aquatics, such as duckweed (*Lemna* spp.), were common in shallow sheltered areas; pondweed (*Potomogeton* spp.) often occurred in deeper water beyond the bulrush margin.

The climate is continental and is characterized by short, cool summers and long, cold winters. Annual precipitation averages 45 cm, 60 percent of which falls as rain between April and October (LWCNRSB 1975b).

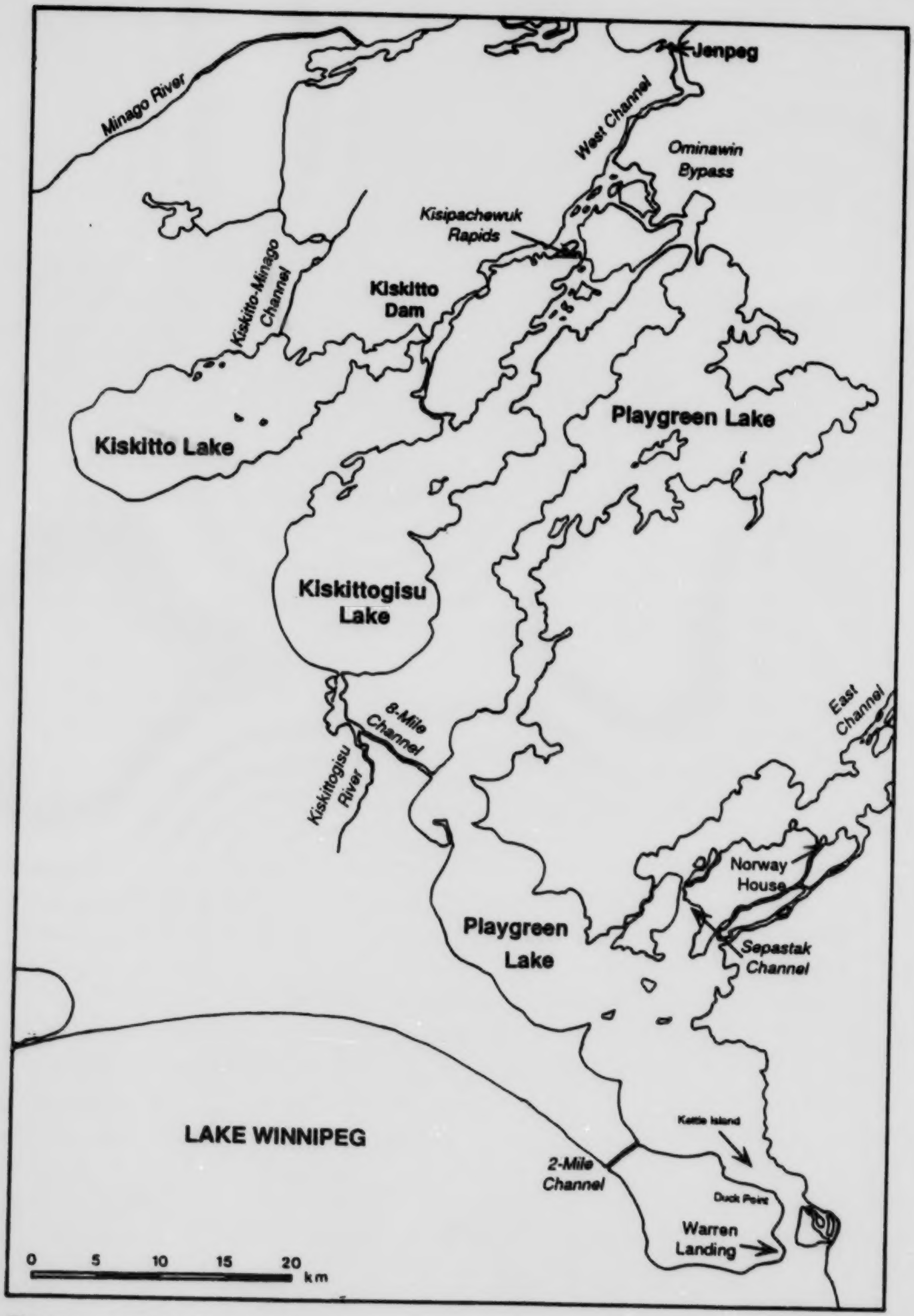
Jenpeg
Minago River
West Channel
Ominawin Bypass
Kisipachewuk Rapids
Kiskitto-Minago Channel
Kiskitto Dam
Kiskitto Lake
Playgreen Lake
Kiskittogisu Lake
8-Mile Channel
Kiskittogisu River
East Channel
Norway House
Sepastak Channel
Playgreen Lake
LAKE WINNIPEG
Kettle Island
2-Mile Channel
Duck Point
Warren Landing
0
5
10
15
20
km

Figure 1. Outlet Lakes study area.

3. LAKE WINNIPEG REGULATION PROJECT

The following details of the LWR project were extracted from the Summary Report of the Lake Winnipeg, Churchill and Nelson Rivers Study (LWCNRSB 1975c), supplemented by other information from Manitoba Hydro.

Regulation of the outflow of Lake Winnipeg is a major component of hydroelectric development on the Nelson River. Before regulation, natural seasonal variations in outflow of Lake Winnipeg were out of phase with power demand. The regulation project was designed to ensure that releases from Lake Winnipeg could be increased during the winter, the higher power demand period, and that water could be stored from one year to the next in the event of low runoff conditions in the drainage basin. To accomplish this, it was necessary to increase the discharge capacity from the Lake and to install a structure across the outlet that would permit flow releases from the Lake to be controlled.

The Jenpeg Generating Station was constructed on the Nelson River West Channel and controls approximately 30 percent of the outflow from Lake Winnipeg (Figure 1). The remaining flow passes down the uncontrolled East Channel through Pipestone Lake into Cross Lake. In addition to controlling flows, Jenpeg can also produce up to 168 mW of power.

Three new channels were excavated to increase the capacity to discharge water from Lake Winnipeg to the Nelson River: 2-Mile Channel, 8-Mile Channel and the Ominawin Bypass (Figure 1). In addition, limited channel improvement work was carried out at Kisipachewuk Rapids. The 2-Mile Channel increased the discharge capacity between Lake Winnipeg and Playgreen Lake, while 8-Mile Channel increased the discharge capacity between Playgreen and Kiskittogisu lakes. The Ominawin Bypass diverts a portion of the flow around Ominawin Rapids and into the West Channel.

The Kiskitto Lake dam, and associated dyke system, is an integral part of the West Channel control system which prevents flows from bypassing the Jenpeg Generating Station (Figure 1). The dam and dykes isolate Kiskitto Lake from the raised water levels in the West Channel. A gated conduit installed in the dam permits flows from the West Channel into Kiskitto Lake to augment natural inflow into the lake, when required. Outflow from the lake is discharged through a stop-log control structure into the Kiskitto-Minago Channel. This channel was constructed to connect Kiskitto Lake to Black Duck Creek and allows Kiskitto Lake outflows to enter the Minago River and Cross Lake.

Implementation of the LWR project began in May 1972 with construction of the Jenpeg tailrace cofferdam. By July 1976, all facilities were sufficiently in place to permit full, unrestricted regulation of Lake Winnipeg for the first time.

4. PRE-PROJECT ENVIRONMENTAL ASSESSMENT

4.1 Lake Winnipeg, Churchill and Nelson Rivers Study

4.1.1 Study Findings

In its investigations of the physical impacts of Lake Winnipeg regulation, LWCNRSB (1975b) predicted that changed water levels would affect marsh zones in the north and south basins of Playgreen Lake. Altered water level regimes were expected to "not only reduce the total width of the marsh zone, but also be instrumental in changing the species composition of the marsh zone".

Water level increases were expected to occur in Kiskittogisu Lake also, resulting in shoreline flooding, particularly in the low-lying southern portion of the lake. Because Kiskitto Lake was to be regulated independently of Lake Winnipeg, its levels were not expected to fluctuate in the same manner as the other lakes. However, a rise in the lake's mean level, combined with reduced water fluctuations, were still expected to cause a substantial reduction of marsh areas.

Webb (1973) noted that fall staging use of Playgreen Lake by scaup and other ducks was moderately high in 1972. Shorelines along the south basin of Playgreen Lake were rated by the Canada Land Inventory as being important areas for migrating waterfowl (Hutchison and Adams 1970). Waterfowl were expected to continue to use the lake as a staging area in fall, but at reduced levels due to Lake Winnipeg regulation. Water level increases were expected to remove some waterfowl nesting habitat.

Moderate numbers of migrating tundra swans, scaup, mallards and Canada geese used Kiskittogisu Lake in fall, 1972. Hutchison and Adams (1970) identified a number of areas that served as important migration stops. Some birds were expected to be displaced as a result of habitat changes caused by modified flow regimes. Numerous nesting islands could also be inundated. Excavation of 8-Mile Channel and associated deposition of dredge spoils was expected to adversely affect the habitat of nesting waterfowl and sandhill cranes and lead to a reduction in use by those species. The channel was expected to carry additional sediment loads to Kiskittogisu Lake and increase its turbidity, which in turn could adversely affect the growth of submergent vegetation and limit the availability of food for staging waterfowl.

Hutchison and Adams (1970) considered parts of the lake to be important for staging waterfowl. The Study Board considered that regulation of Kiskitto Lake levels independent of West Channel flows would result in less frequent flooding of nesting islands in the lake and reduce the number of waterfowl nests flooded in the spring. However, if water levels were maintained within too narrow a range, stagnation and loss of marsh vegetation could occur and the lake's value to waterfowl would be reduced.

4.1.2 Study Board Recommendations

The Study Board made a number of recommendations that concerned the entire Nelson River hydroelectric development scheme, and others that were specific to the Outlet Lakes area (LWCNRSB 1975c).

Recommendations of general application that are relevant to the present study as follows:

> 5. *That a mechanism be established to deal with social and related and economic issues including:*
>
>   (c) *monitoring and analysis of ongoing social and economic changes related to hydroelectric development and more generally northern development.*
>
> 10. *That appropriate government departments and agencies develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the affected area.*

One recommendation, specific to the Outlet Lakes area, applies directly to the present study:

29. *That resource management and resource agencies investigate further the operational aspects of the Kiskitto Lake regulation works to determine optimum water levels for fish, wildlife and recreational resources.*

4.2 Canadian Wildlife Service Studies

4.2.1 Kiskittogisu River

The impact of construction of 8-Mile Channel on migratory bird use of Kiskittogisu River and adjacent habitat was assessed by Rakowski (1975).

Kiskittogisu River, including its lagoon system at the downstream end, was considered to provide good quality rearing habitat for ducks and geese and was regarded as important for staging and migrating waterfowl. This habitat was expected to be severely affected by sedimentation as a result of deposition of dredge spoils during construction of the 8-Mile Channel. Rakowski predicted that all available high-quality waterfowl habitat in the vicinity of Kiskittogisu River would be lost through flooding once the channel connected Playgreen and Kiskittogisu lakes.

4.2.2 Kiskitto Lake

In his report to the Study Board, Webb (1973) recommended that optimal water levels for Kiskitto Lake be determined to lessen the impact of Lake Winnipeg regulation on wildlife. Subsequently, Canadian Wildlife Service initiated a study to evaluate the current and potential value of Kiskitto Lake to waterfowl, and to recommend development and management practices that would benefit waterfowl.

Rakowski (1976) concluded that the lake was probably more important for waterfowl staging than production, and recommended that lake levels be maintained between 212.60 and 213.36 m a.s.l. The water level should be lowered in the spring and maintained at approximately 212.60 m a.s.l. until August to (a) create additional shoreline habitat for nesting waterfowl; (b) prevent flooding of waterfowl nests; and (c) maintain habitat in optimum condition. High water levels could be maintained every third or fourth year in order to control undesirable plant succession.

4.3 Kiskitto Lake Regulation Committee Study

The Kiskitto Lake Regulation Committee, comprised of representatives of the Manitoba government and Manitoba Hydro, was formed in August 1975 in response to Study Board Recommendation 29 (see subsection 4.1.2). The terms of reference of the Committee required it to "develop operating rules for the Kiskitto Lake control structure that will optimize the value of the resources associated with the lake, recognizing environmental and social considerations in addition to the economic value of the resources" (Kiskitto Lake Regulation Committee 1977).

The Committee recommended that the lake be regulated between 212.75 and 213.36 m a.s.l., and that a minimum inflow of 2.83 m^3/sec from the Nelson River West Channel be maintained throughout the year to prevent eutrophication. To improve flushing of the lake, inflow should be increased to full culvert capacity when water is being spilled at the Jenpeg Generating Station. However, lake levels should not be permitted to exceed the 213.36m upper elevation to prevent flooding of muskeg.

5. METHODS

5.1 Impact Assessment and Impact Management Approach

The flowchart in Figure 2 illustrates the approach used in determining how the waterfowl-related interests and opportunities of Norway House Band Indians in the fall have been affected by the LWR project and, consequently, what compensation is required by the NFA. References are made in the flowchart to the pertinent recommendations of the Study Board and provisions of the NFA which are addressed by the process. Reference to "adverse" effects or impacts in this report reflects terminology of the NFA and does not represent any preconclusion of this study.

The assessment process begins with identifying the effects of LWR on waterfowl use of the Outlet Lakes area in fall (biophysical impacts). The direct, indirect, or cumulative consequences of these biophysical impacts are expected to affect the economic, cultural, and social life of Norway House Band people.

5.1.1 Biophysical Impacts

Determining how the LWR project has affected waterfowl use of the Outlet Lakes area partially fulfills Recommendation 10 of the Study Board and NFA sub-article 17.5 (See Appendix A).

Recommendation 10 recognizes that ongoing management of the Outlet Lakes area requires that information first be gathered on how the area has been impacted by development. The present study focuses on how use of the Outlet Lakes by fall-staging waterfowl has been affected by the Lake Winnipeg regulation.

NFA sub-article 17.5 (of Article 17 - Environmental Impact Policy) requires that monitoring of effects of the project be carried out so as to "provide such information as may be necessary to give effect" to the NFA (see Appendix A). Giving effect to the NFA means ensuring that all members of the five NFA

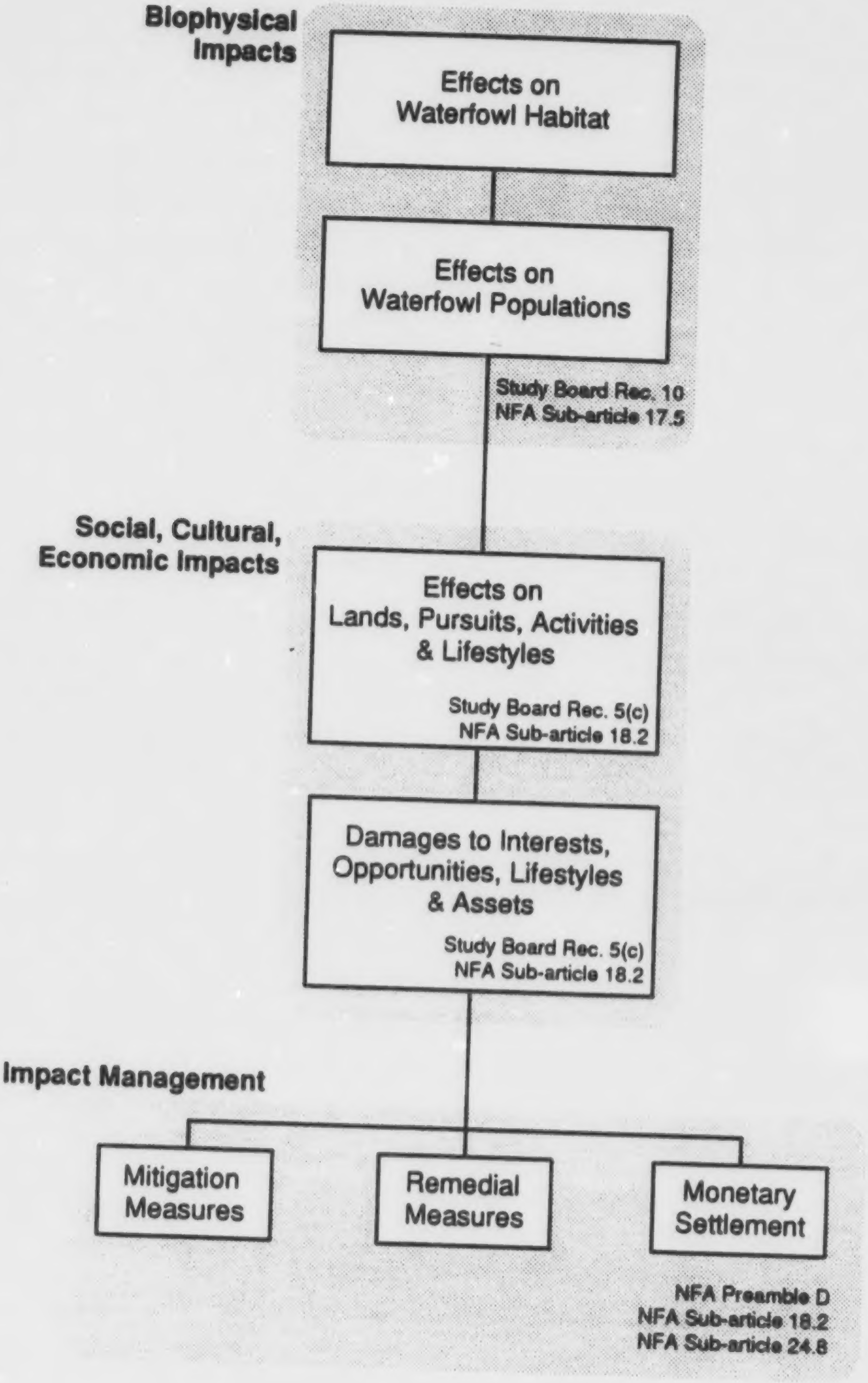
Biophysical Impacts
Effects on Waterfowl Habitat
Effects on Waterfowl Populations
Study Board Rec. 10
NFA Sub-article 17.5
Social, Cultural, Economic Impacts
Effects on Lands, Pursuits, Activities & Lifestyles
Study Board Rec. 5(c)
NFA Sub-article 18.2
Damages to Interests, Opportunities, Lifestyles & Assets
Study Board Rec. 5(c)
NFA Sub-article 18.2
Impact Management
Mitigation Measures
Remedial Measures
Monetary Settlement
NFA Preamble D
NFA Sub-article 18.2
NFA Sub-article 24.8

Figure 2. The impact assessment and impact management approach used in the study.

who have been or may be adversely affected by the hydroelectric development, are dealt with or compensated "fairly and equitably" (Preamble D to the NFA).

Collection of data that document the effects of LWR on waterfowl use of the Outlet Lakes constitutes compliance with sub-article 17.5 with respect to the waterfowl-related impacts that the Norway House claims to have experience. In addition, documentation of effects on waterfowl in the Outlet Lakes area represents a response, in part, to Part III of Claim 18, which embodies the intent and meaning of Recommendation 10 and NFA sub-article 17.5. Claim 18 is described in Appendix A.

5.1.2 Socio-Cultural-Economic Impacts

Determining how LWR has affected the waterfowl-related interests of Norway House Band people partially fulfills Study Board Recommendation 5(c) and NFA sub-article 18.2.

Recommendation 5(c) called for "monitoring and analysis of ongoing social and economic changes related to hydroelectric development" (subsection 4.1.2 of this report). The present study addresses those changes experienced by Norway House Band people that are associated with effects on waterfowl use of the Outlet Lakes area, i.e., changes in waterfowl hunting opportunities.

Collection of information that documents the effects of LWR on the waterfowl hunting opportunities of Norway House Band people constitutes compliance with NFA sub-article 18.2 with respect to the waterfowl-related impacts experienced by the Norway House Band. In addition, this documentation represents a response, in part, to Part II of Claim 18, which embodies the intent and meaning of Recommendation 5(c) and NFA sub-article 18.2. Finally, information dealing with adverse effects on waterfowl hunting opportunities allows damages to the waterfowl-related interests, opportunities, lifestyles and assets of Norway House people to be defined.

5.1.3 Impact Management

Impact Management refers to a mechanism, or set of mechanisms, by which adverse impacts of a project may be managed, whether they be biophysical (e.g., directly affecting waterfowl) or socio-cultural-economic impacts (e.g., affecting waterfowl hunting opportunities). Suitable and adequate compensation for adverse effects on NFA Bands would constitute redress for damages suffered by NFA Bands. NFA sub-article 24.8 refers to three forms of compensation or redress for damages suffered: mitigating measures, remedial measures and monetary settlement.

5.2 Data Collection

5.2.1 Effects on Waterfowl and Habitat

On June 20, 1986, an aerial reconnaissance of the 8-Mile Channel and Kiskittogisu River area was carried out using a Bell 206B helicopter to determine general habitat conditions following construction of 8-Mile Channel. Additional observations on shoreline habitat were made during the aerial surveys of waterfowl populations conducted in fall, 1986 and 1987.

Fall staging of waterfowl in the Outlet Lakes area was determined through a series of low-altitude aerial surveys conducted in 1986 and 1987. The 1986 surveys were flown on September 5 and 12 in a Cessna 185 with two observers; the October 5, 16 and 26, 1987 surveys were flown in a Bell 206B helicopter with one observer. All Surveys were flown at altitudes of between 30 and 40 m above ground level. Survey routes followed shorelines of Playgreen, Kiskitto and Kiskittogisu lakes and 8-Mile Channel (Figure 3). Table 1 gives the length of each survey in each lake.

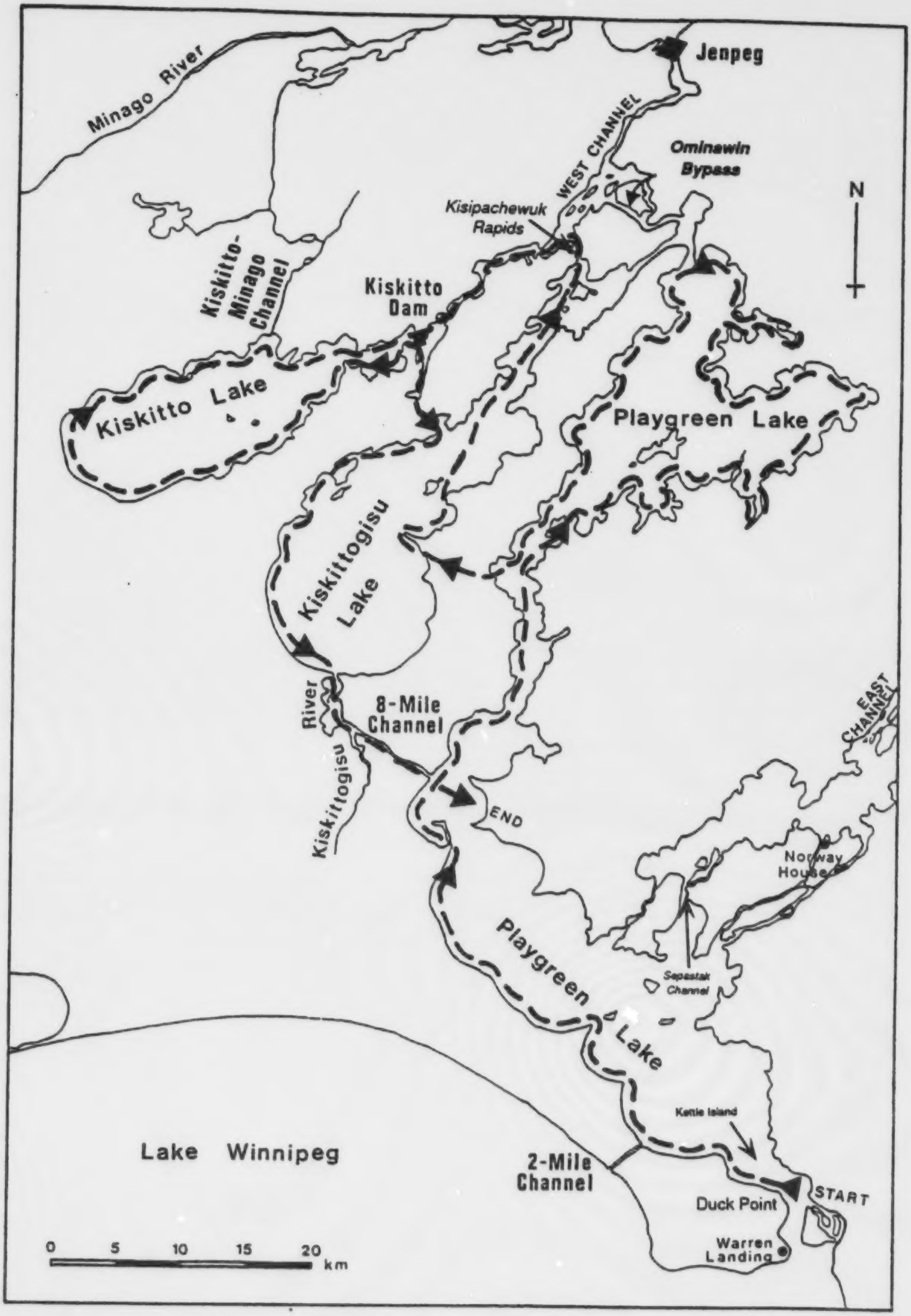
Jenpeg
Minago River
WEST CHANNEL
Ominawin Bypass
N
Kisipachewuk Rapids
Kiskitto-Minago Channel
Kiskitto Dam
Kiskitto Lake
Playgreen Lake
Kiskittogisu Lake
River
Kiskittogisu
8-Mile Channel
END
EAST CHANNEL
Norway House
Sepastak Channel
Playgreen Lake
Kettle Island
START
Lake Winnipeg
2-Mile Channel
Duck Point
Warren Landing
0
5
10
15
20
km

Figure 3. Waterfowl survey route, 1986/87.

Table 1. Lengths (km) of post-regulation surveys of the Outlet Lakes area.

Area	Sept. 5, 1986	Sept. 12, 1986	Oct. 5, 1987	Oct. 16, 1987	Oct. 26, 1987
Playgreen-South Basin	59.0	59.0	137.0	145.0	133.0
Playgreen-North Basin	151.0	151.0	151.0	145.5	145.5
Kiskitto Lake	85.0	85.0	85.0	30.8	n.s.
Kiskittogisu Lake	101.0	101.0	101.0	12.0	13.5
TOTAL	396.0	396.0	474.0	333.3	292.0

n.s. = not surveyed

5.2.2 Effects on Norway House Band

On February 13, 1987, one-on-one interviews were held with seven waterfowl hunters in Norway House. Hunters were asked to describe: their waterfowl hunting experiences before and after Lake Winnipeg regulation; traditional hunting locations used prior to regulation; hunting success at traditional hunting locations before and after regulation; and species of waterfowl hunted. They were also asked to share their observations on changes in numbers and distribution of waterfowl and their perceptions on the causes of these changes and the influence of Lake Winnipeg regulation. Because the purpose of the study was to investigate the effects of the diversion on fall waterfowl use and fall hunting, discussions focussed on the fall hunting period.

6. RESULTS

6.1 General Habitat Conditions

Habitat conditions along Kiskittogisu River and in the vicinity of 8-Mile Channel were examined by helicopter on June 20, 1986. Figure 4 shows how 8-Mile Channel intercepts the former natural outlet of Kiskittogisu River before entering Kiskittogisu Lake.

8-Mile Channel has caused the level of Kiskittogisu River to exceed its natural capacity and flood vegetation that previously grew along its banks. Remains of dead white birch (*Betula papyrifera*), black spruce (*Picea mariana*) and other trees occur along the river course.

Although no ground inspection was carried out, the islands created by depositing dredge spoils along 8-Mile Channel have become well-vegetated with willow (*Salix*) and various aquatic plants. Where Kiskittogisu River enters 8-Mile Channel appears to have become stabilized with aquatic vegetation.

6.2 Waterfowl Use

Data on waterfowl use have been summarized for the following survey areas: Playgreen Lake north basin, Playgreen Lake south basin, Kiskitto Lake and Kiskittogisu Lake (Figure 1).

The survey results presented here focus primarily on numbers of birds observed per kilometre surveyed. Because survey dates are widely separated both within and between years, and represent different observation "platforms" (fixed-wing and helicopter aircraft), no attempt has been to calculate average densities. Instead, discussion focuses on the highest densities recorded because they are more likely to represent the potential for waterfowl use. Highest densities are also used in the comparison of pre- and post-regulation use of the Outlet Lakes by waterfowl (see Section 7).

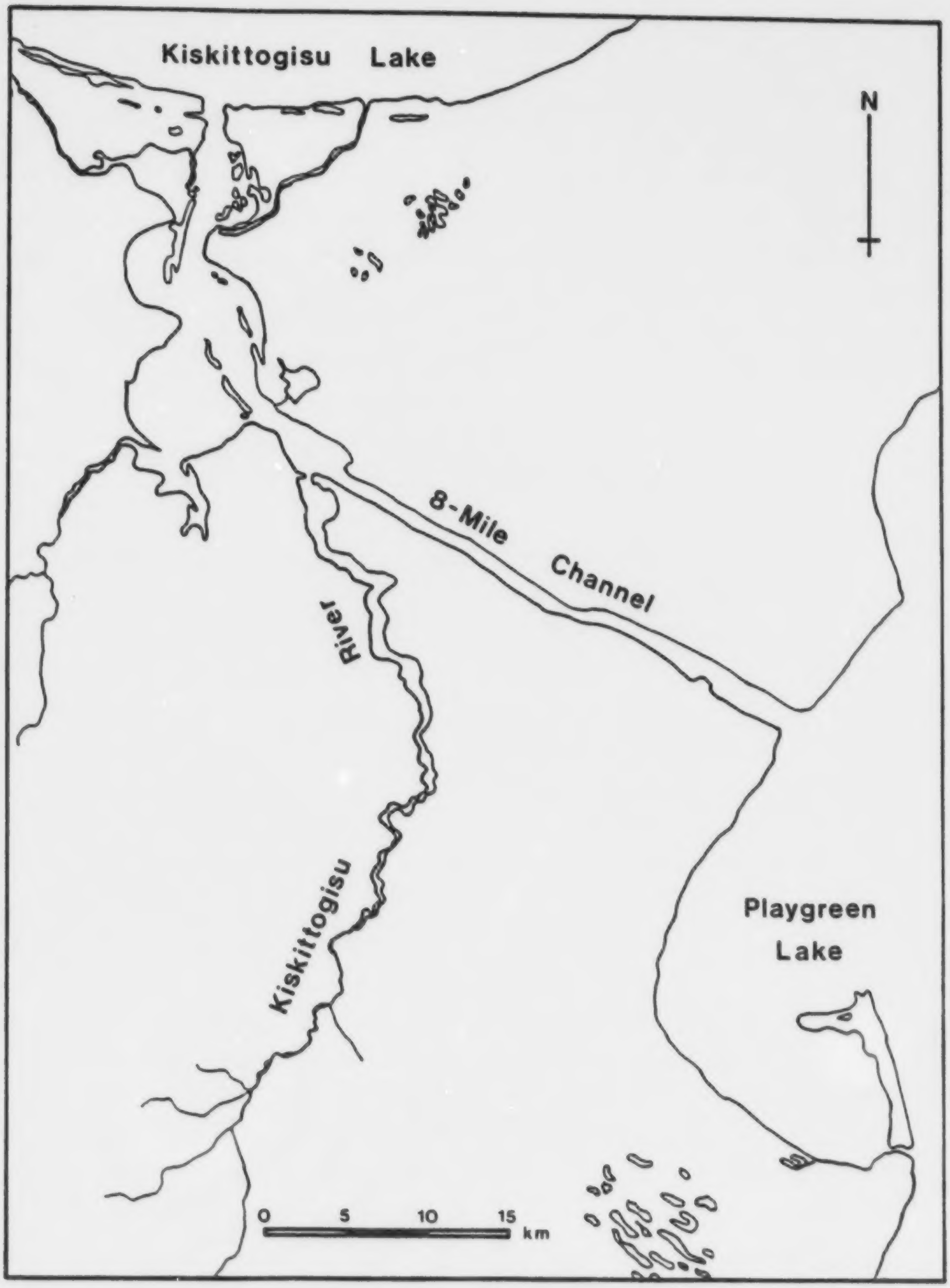
Kiskittogisu Lake
N
8-Mile Channel
River
Kiskittogisu
Playgreen Lake
0
5
10
15
km

Figure 4. Location of 8-mile channel in relation to Kiskittogisu River, Kiskittogisu Lake, and Playgreen Lake (south basin).

The September and October survey dates were chosen to: (a) correspond with pre-regulation surveys carried out in 1972 and 1973, and (b) coincide with major waterfowl movements based on long-term experience of Norway House residents.

Diving Ducks

Playgreen Lake received the most use by diving ducks in 1986/87 with the north basin attracting more birds per kilometre of shoreline than the south basin (Figure 5). Lesser scaup (*Aythya affinis*) was the most abundant diver species while common goldeneye (*Bucephala clangula*), bufflehead (*Bucephala albeola*) and scoters (*Melanitta* spp.) were observed in lower numbers.

Highest densities of diving ducks recorded during any survey segment of the Outlet Lakes ranged from .13 birds/km at Kiskitto Lake to 2.69 birds/km at Playgreen Lake, North Basin; densities at Playgreen Lake, South basin were almost as high -- 2.37 birds/km (Table 2, Figure 5). Densities were consistently higher in 1987 than in 1986, which probably reflects the difference in survey techniques for the two years (fixed-wing aircraft in 1986; helicopter in 1987). Two of the Kiskitto Lake surveys and two of the Kiskittogisu Lake surveys yielded no diving ducks.

Dabbling Ducks

Dabbling duck use of Playgreen Lake was low in 1986 (Table 3, Figure 5). Mallards (*Anas platyrhynchos*) were the most common frequently observed species, with gadwall (*Anas strepera*) and American wigeon (*Anas americana*) present in lower numbers.

Highest densities of dabbling ducks during individual survey segments varied from 0.30 birds/km (Kiskitto Lake) to 4.58 birds/km (Kiskittogisu Lake). The highest density recorded at Playgreen Lake, North Basin was 3.06 birds/km, versus 2.50 recorded in the South Basin. Ducks were observed on all but one survey (September 12, 1986) of the two Playgreen Lake segments; however, at

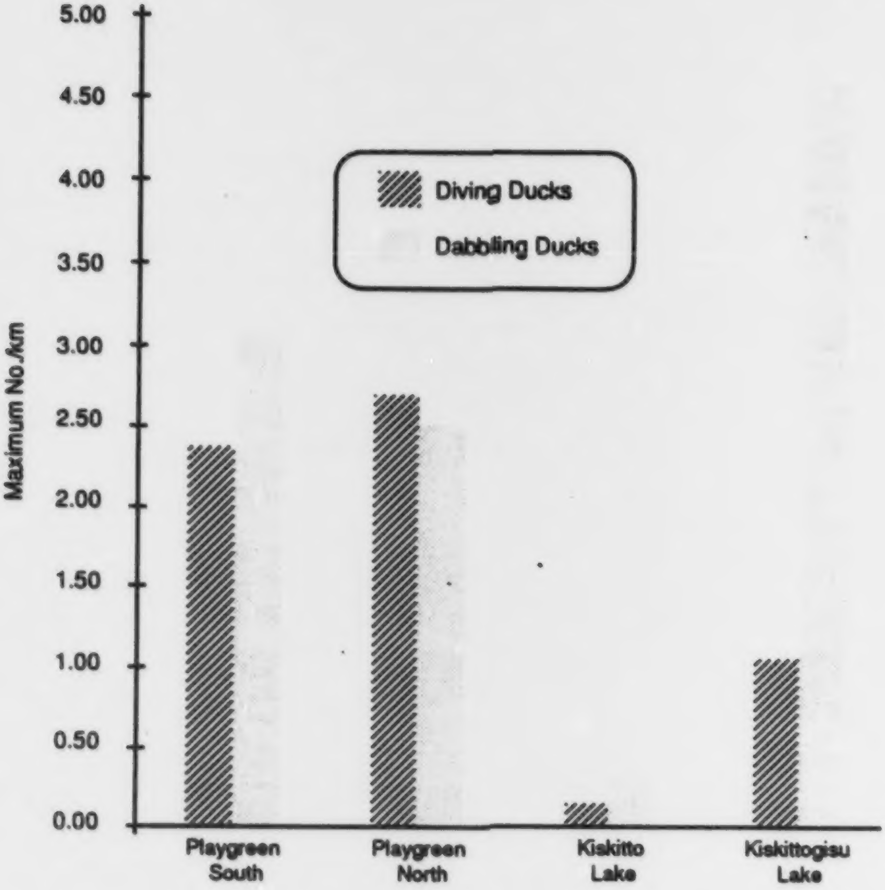

Figure 5. Diving and dabbling duck use of the Outlet Lakes area in Fall, 1986/87.

Table 2. Numbers and densities* of diving ducks observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986		12 Sept. 1986		5 Oct. 1987		16 Oct. 1987		26 Oct. 1987		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	0	0.00	4	0.07	92	0.67	212	1.46	315	2.37	2.37
Playgreen–North Basin	309	2.05	99	0.66	406	2.69	308	2.12	287	1.97	2.69
Kiskitto Lake	10	0.12	0	0.00	0	0.00	4	0.13	n.s.	–	0.13
Kiskittogisu Lake	25	0.25	33	0.33	105	1.04	0	0.00	0	0.00	1.04
TOTAL & AVG. DENSITY	**344**	**0.87****	**136**	**0.34****	**603**	**1.27****	**524**	**1.57****	**602**	**2.06****	

* birds/km

**Based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)

n.s. = not surveyed

Table 3. Numbers and densities* of dabbling ducks observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986		12 Sept. 1986		5 Oct. 1987		16 Oct. 1987		26 Oct. 1987		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	2	0.03	0	0.00	163	1.19	444	3.06	32	0.24	3.06
Playgreen–North Basin	91	0.60	16	0.11	103	0.68	364	2.50	14	0.10	2.50
Kiskitto Lake	0	0.00	0	0.00	25	0.30	0	0.00	n.s.	–	0.30
Kiskittogisu Lake	30	0.30	0	0.00	137	1.36	55	4.58	0	0.00	4.58
TOTAL & AVG. DENSITY	**123**	**0.31****	**16**	**0.04****	**428**	**0.9****	**863**	**2.59**	**46**	**0.16****	

* birds/km

**based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)

n.s. = not surveyed

Kiskitto Lake birds were observed on only one of the surveys (October 5, 1987). The high density recorded at Kiskittogisu Lake (October 16, 1987) is based on a short survey segment (12 km) and may represent a biased estimate of waterfowl use for the entire lake basin.

Ducks (Dabbling, Diving, and Unidentified species)

Highest densities for all ducks (Table 4, Figure 6) were about the same for Playgreen Lake North (4.86 birds/km) and South (4.62 birds/km) Basins and for Kiskittogisu Lake (4.58 birds/km). Given the range of time covered by the surveys -- early September to late October over two seasons -- the densities for Playgreen North and, to a lesser extent, Playgreen South are surprisingly consistent. Results also suggest that the October 16 survey best represented ducks present in the Outlet Lakes region in Fall, 1987. The generally low densities recorded in 1986 may indicate surveys that year occurred before large numbers of migrants had reached the study area.

Canada Geese

Numbers and densities of Canada geese were low through all survey dates and segments (Table 5, Figure 6), with highest densities ranging from 0.19 birds/km (Playgreen Lake North Basin) to 1.46 birds/km (Playgreen Lake South Basin). The largest number of geese recorded on any survey segment/date was 54.

6.3 Hunter Interviews

Traditionally, Norway House Band members have hunted waterfowl during spring and fall migrations. Canada geese are hunted almost exclusively in spring, whereas ducks are hunted during both seasons.

Waterfowl hunters expressed their concerns about the effects of diversion during interviews conducted as part of this study (Figure 7). Hunters felt that the availability of ducks and geese at traditional hunting sites is considerably lower than it was before LWR. As a consequence, hunters were returning home with

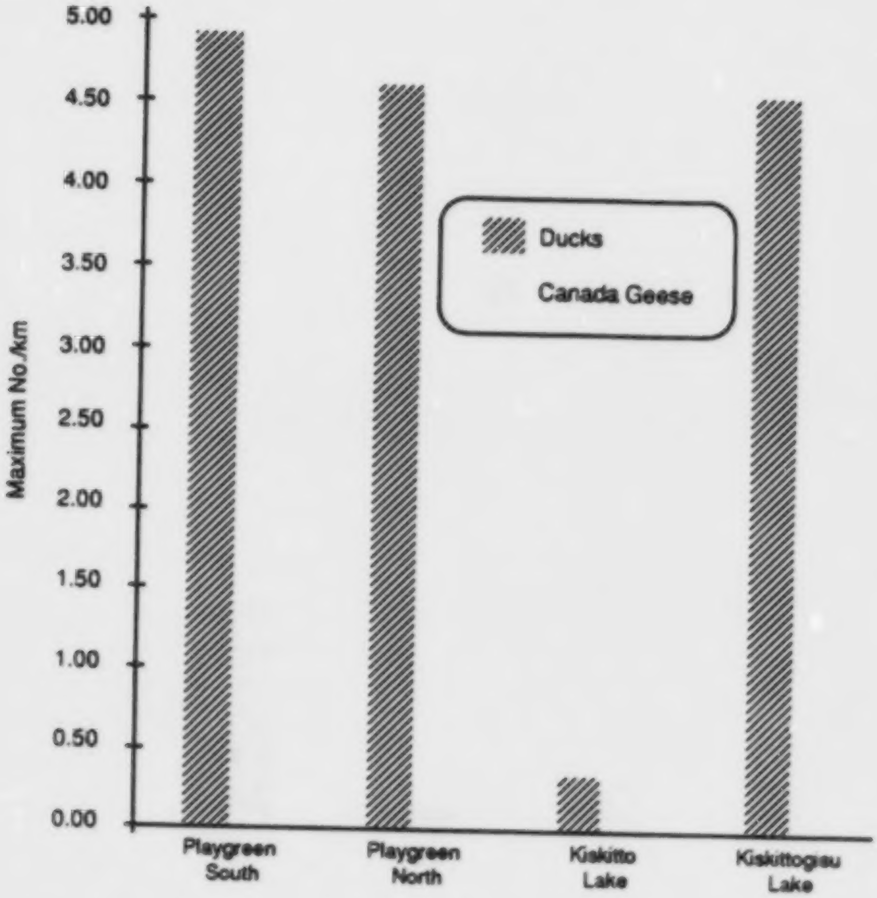

Figure 6. Use of the Outlet Lakes area by ducks (divers, dabblers, unidentified) and Canada geese in Fall, 1986/87.

WATERFOWL-RELATED CONCERNS AND ADVERSE EFFECTS

Observations of Norway House Band Hunters*

Habitat

- pondweeds are not as numerous as before
- pondweeds seem to be drowned out
- Playgreen Lake has become murky
- there are concerns about mercury levels and organic material in the lake

Waterfowl

- ducks generally are not as numerous as before regulation
- ducks are not as numerous in the Warren Landing narrows
- there has been a noticeable drop in duck numbers in the Sepastak Channel area
- ducks tend to raft in Playgreen Lake away from shore; they used to be distributed in shallow bays close to shore
- there has been no noticeable change in numbers of ducks and Canada geese in spring
- reductions in duck numbers have occurred since 2-mile Channel and the transmission line were installed
- most ducks now go as far as Kettle Island or Duck Point then rise to a higher altitude; they used carry on in low flights to Warren Landing Narrows

Hunting

- there has been little success in hunting at traditional sites; returns have been poor for the effort spent
- it used to be easy to decoy ducks to Kettle Island before regulation
- there is concern over the size of the hunter-kill in the United States
- hunters used to be able to rely on a succesful hunt near Warren Landing and along the east shore of Playgreen Lake
- Kiskittogisu River used to be good for geese in spring and fall
- hunters used to be able to bag about 10-15 ducks an hour; four years ago, they bagged only 3 ducks over three evenings

*Comments listed here represent the opinions of individual hunters

Figure 7. Waterfowl-related concerns and adverse effects identified during hunter interviews at Norway House.

Table 4. Numbers and densities* of ducks^ observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986		12 Sept. 1986		5 Oct. 1987		16 Oct. 1987		26 Oct. 1987		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	2	0.03	4	0.07	310	2.26	705	4.86	382	2.37	4.86
Playgreen–North Basin	400	2.65	107	0.71	559	3.70	672	4.62	301	2.07	4.62
Kiskitto Lake	10	0.12	0	0.00	25	0.30	4	0.13	n.s.	–	0.30
Kiskittogisu Lake	55	0.55	33	0.33	242	2.40	55	4.58	0	0.00	4.58
TOTAL & AVG. DENSITY	467	1.18**	144	0.36**	1136	2.4**	1436	4.31**	683	2.34**	

* birds/km
^includes dabblers, divers, and unidentified ducks
**based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)
n.s. = not surveyed

Table 5. Numbers and densities* of Canada geese observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986		12 Sept. 1986		5 Oct. 1987		16 Oct. 1987		26 Oct. 1987		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	0	0.00	0	0.00	28	0.67	17	1.46	0	0.00	1.46
Playgreen–North Basin	15	0.10	29	0.19	17	0.11	23	0.16	0	0.00	0.19
Kiskitto Lake	8	0.10	54	0.64	0	0.00	0	0.00	n.s.	–	0.64
Kiskittogisu Lake	38	0.38	15	0.15	42	0.42	0	0.00	0	0.00	0.42
TOTAL & AVG. DENSITY	61	0.15**	98	0.25**	87	0.18**	40	0.12**	0	0.00	

* birds/km
**based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)
n.s. = not surveyed

substantially fewer ducks per unit effort than they harvested before regulation. In contrast, spring waterfowl hunting opportunities appeared to remain largely unchanged.

Figure 8 shows the areas frequently used for hunting ducks in the fall prior to LWR. Of these, the group of islands in the vicinity of Warren Landing received the most use. Hunters described how, in the past, they observed flocks of ducks flying southeast in the fall across Playgreen Lake to the Warren Landing area en route to Lake Winnipeg and points south. Hunters would set up blinds on the shores of the islands east of Warren Landing and decoy ducks into shallow bays where they could be shot and retrieved relatively easily.

Hunters complained that since 2-Mile Channel was constructed and Lake Winnipeg regulation began, ducks no longer visit the Warren Landing area in large numbers. Consequently, hunters get very few geese for their hunting effort. Some hunters observed ducks deviating from their traditional flight path towards Warren Landing and using 2-Mile Channel to gain access to Lake Winnipeg. They indicated that the deep and fast-flowing waters of 2-Mile Channel are not suitable for hunting with decoys. Other hunters had seen flocks travel as far as Kettle Island or Duck Point (Figure 8) and raft offshore in waters too deep to permit recovery of birds. Similarly, hunters reported noticeable reductions in the number of ducks in the "Sepastak" area, and in the bays along the eastern shoreline of Playgreen Lake.

Some hunters noted physical changes in Playgreen Lake and claimed the lake was now more turbid.

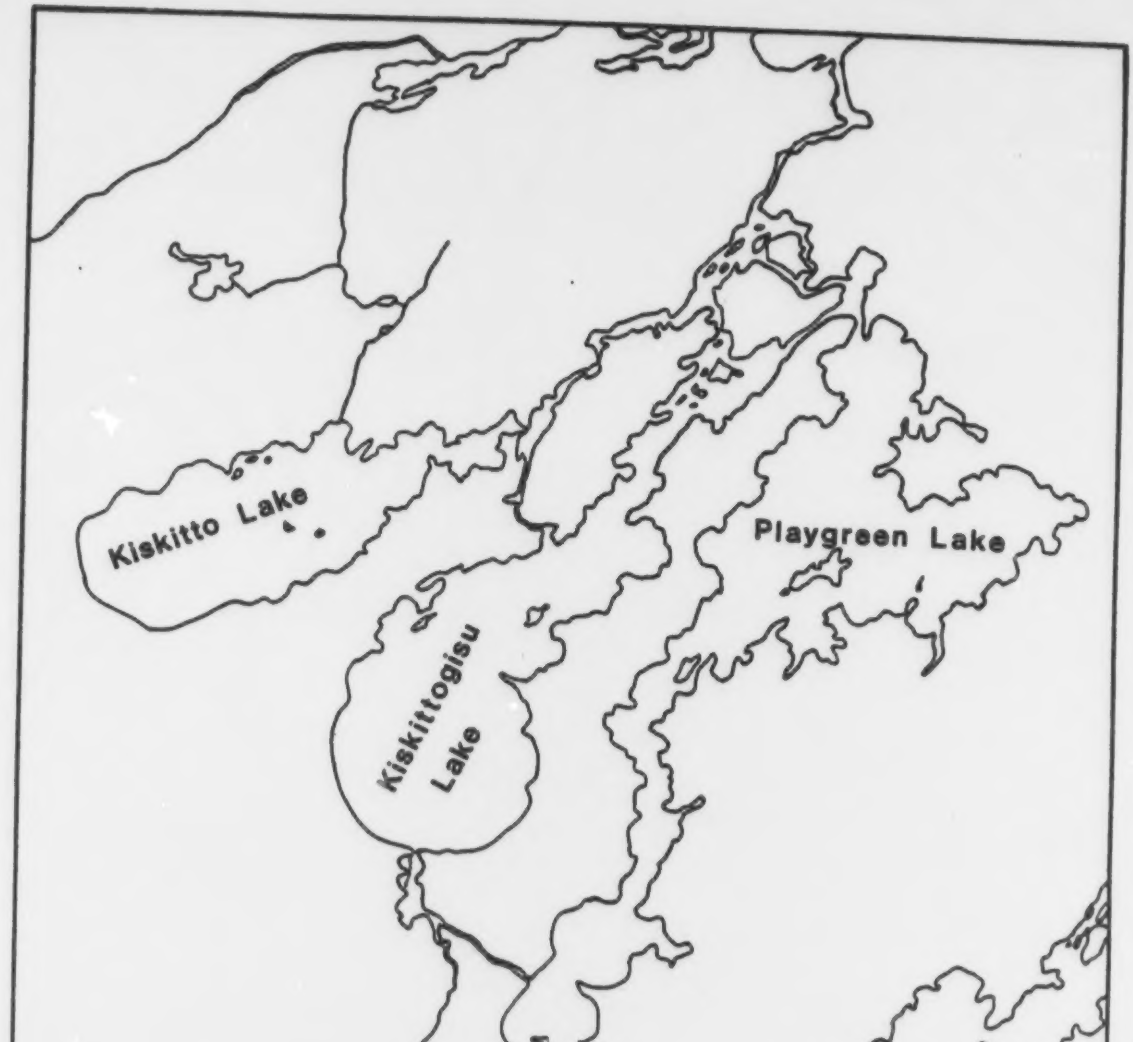
Kiskitto Lake
Kiskittogisu Lake
Playgreen Lake
Playgreen Lake

Figure 8. Traditional fall waterfowl hunting areas of the Norway House Band in the Outlet Lakes area.

7. DISCUSSION

7.1 Pre- and Post-Regulation Waterfowl Use

Aerial surveys of shorelines in the Outlet Lakes area were carried out before Lake Winnipeg regulation by Webb (1973) on September 24 and 25, 1972 and by Manitoba government biologists on August 30, September 7, September 14 and October 6, 1973 (Rakowski 1976). Table 6 gives the length of each survey in each lake.

Diving Ducks

Comparison of 1972/73 and 1986/87 survey results shows densities of diving ducks in the two Playgreen Lake basins to be substantially higher before LWR (Table 7) than after (Table 2). Highest densities of 72.07 and 102.01 birds/km recorded in Playgreen Lake north and south basins before regulation were 30 to 40 times greater than highest densities (2.37 and 2.69 birds/km) reported after regulation.

Densities of diving ducks at Kiskitto and Kiskittogisu lakes were also greater before LWR, although the data for Kiskitto Lake are too fragmentary for firm conclusions to be drawn. The highest density at Kiskittogisu Lake in 1972/73 (4.83 birds/km) was about 5 times greater than that recorded in 1986/87 (1.04 birds/km).

Dabbling Ducks

Comparison of pre-and post-regulation survey results (Tables 8 and 3) for dabblers is inconclusive. In 1972/73, dabbling ducks were observed on Playgreen Lake South Basin on three of five surveys, and on the North Basin on only two of five surveys. In 1986/87, birds were consistently observed on Playgreen Lake through the entire period surveyed. The highest density recorded for South Basin were about 2 times greater in 1986/87 than in 1972/73 (3.06 versus 1.49 birds/km). In North Basin, the highest pre-regulation density was 3 time greater than the highest post-regulation density (7.87 versus 2.5 birds/km). Overall, there

Table 6. Lengths (km) of pre-regulation surveys of the Outlet Lakes area.

Area	Sept. 24/25 1972	Aug. 30 1973	Sept. 7 1973	Sept. 14 1973	Oct. 6 1973
Playgreen-South Basin	113.5	88.8	88.8	19.5	59.0
Playgreen-North Basin	71.8	49.8	n.s.	94.0	n.s.
Kiskitto Lake	85.8	36.5	57.0	27.4	84.0
Kiskittogisu Lake	154.8	45.1	54.6	45.0	92.5
TOTAL	425.9	220.2	200.4	185.9	235.5

n.s. = not surveyed

Table 7. Numbers and densities* of diving ducks observed during pre-regulation surveys of the Outlet Lakes area.

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	2930	25.81	4135	46.57	6400	72.07	277	14.21	1505	25.51	72.07
Playgreen–North Basin	4534	63.15	5080	102.01	n.s.	n.s	770	8.19	n.s.	n.s.	102.01
Kiskitto Lake	174	2.03	0	0.00	0	0.00	0	0.00	0	0.00	2.03
Kiskittogisu Lake	747	4.83	85	1.88	0	0.00	0	0.00	300	3.24	4.83
TOTAL & AVG. DENSITY	**8385**	**19.69****	**9300**	**41.85****	**6400**	**31.94****	**1047**	**5.63****	**1805**	**7.66****	

* birds/km

**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).

n.s. = not surveyed

Table 8. Numbers and densities* of dabbling ducks observed during pre-regulation surveys of the Outlet Lakes area.

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	169	1.49	0	0.00	10	0.11	0	0.00	50	0.85	1.49
Playgreen–North Basin	565	7.87	10	0.20	0	0.00	0	0.00	n.s.	n.s.	7.87
Kiskitto Lake	77	0.90	0	0.00	0	0.00	21	0.77	0	0.00	0.90
Kiskittogisu Lake	88	0.57	85	1.88	0	0.00	21	0.47	0	0.00	1.88
TOTAL & AVG. DENSITY	**899**	**2.11****	**95**	**0.43****	**10**	**0.05****	**42**	**0.23****	**50**	**0.21****	

* birds/km

**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).

n.s. = not surveyed

is no indication that use by dabbling ducks has changed since LWR. The survey results do suggest that the Outlet Lakes did not attract large numbers of dabblers, with or without regulation.

As noted for diving ducks, not enough dabbling ducks were observed on Kiskitto Lake surveys to determine the effects of LWR on duck use or populations trends.

Surveys at Kiskittogisu Lake suggest that dabbling duck densities may have increased slightly since regulation; however, the counts are too small and inconsistent both years to draw reliable conclusions. Furthermore, the 1986/87 highest density of 4.58 birds/km is suspect, given the short distance surveyed (12 km).

Ducks (Dabbling, Diving, Unidentified)

As one might expect, the diving duck counts dominate the total-duck picture, which suggests that duck use has declined significantly on Playgreen Lake since LWR (Tables 4 and 9). There is some indication that densities at Kiskittogisu Lake have also declined since LWR, based on a highest density of 8.39 birds/km in 1972/73 versus 4.58 birds/km in 1986/87.

Inclusion of unidentified ducks to the survey results shows total duck densities at Kiskitto Lake to be higher before LWR than after, although numbers again are too small and inconsistent to be conclusive.

Canada Geese

Densities of Canada geese on Playgreen Lake North Basin, Kiskitto Lake, and Kiskittogisu Lake in 1972/73 were 2.6 and 4.1 times greater than in 1986/87, based on highest densities recorded (Tables 10 and 5). The opposite was true for Playgreen Lake South Basin, where goose densities were 7.3 times higher in 1986/87. Given the relatively small numbers of birds observed on individual surveys (12 to 175 in 1972/73; 8 to 42 in 1986/87) and the fact that no geese were

Table 9. Numbers and densities* of ducks^ observed during pre-regulation surveys of the Outlet Lakes area.

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	3315	29.21	4225	47.58	6410	72.18	277	14.21	1555	26.36	72.18
Playgreen–North Basin	5099	71.02	5190	104.22	n.s.	n.s	800	8.51	n.s.	n.s.	104.22
Kiskitto Lake	341	3.98	20	0.55	0	0.00	21	0.77	0	0.00	3.98
Kiskittogisu Lake	1299	8.39	145	3.22	0	0.00	21	0.47	300	3.24	8.39
TOTAL & AVG. DENSITY	10054	23.61**	9580	43.51**	6410	31.99**	1119	6.02**	1855	7.66**	

* birds/km
^includes dabblers, divers, and unidentified ducks.
**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).
n.s. = not surveyed

Table 10. Numbers and densities* of Canada geese observed during pre-regulation surveys of the Outlet Lakes area.

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	0	0.00	10	0.11	18	0.20	0	0.00	12	0.20	0.20
Playgreen–North Basin	56	0.78	22	0.44	n.s.	n.s	13	0.14	n.s.	n.s.	0.78
Kiskitto Lake	87	1.01	97	2.66	12	0.21	0	0.00	0	0.00	2.66
Kiskittogisu Lake	175	1.13	12	0.27	12	0.22	23	0.51	42	0.45	1.13
TOTAL & AVG. DENSITY	318	0.71**	141	0.52**	42	0.21**	36	0.18**	54	0.23**	

* birds/km
**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).
n.s. = not surveyed

observed on several surveys both before and after regulation, it is impossible to say whether the data represent any real change in goose use since LWR.

Figures 9 to 12 and Table 11 show the percentage change in diving and total duck numbers that occurred from 1972/73 (pre-regulation) to 1986/87 (post-regulation) for each of the four survey areas. Data for dabbling ducks and Canada geese are not included in Table 11 summary because the survey results are insufficient to make meaningful comparisons between pre- and post-regulation use by those species.

Results indicate that diving ducks declined by about 97% in Playgreen Lake and 78.5% in Kiskittogisu Lake between pre-and post-regulation periods. While comparisons are included for Kiskitto Lake, they must be considered inconclusive, owing to the limited amount of survey data available.

Considering the Outlet Lakes as a whole (Table 12), one can conclude that use by diving ducks has declined substantially (-96.2%) since LWR. The large decline (-90.1%) indicated for all ducks largely reflects the large diving duck component.

In general, it is risky to draw broad conclusions about changes in animal numbers from two periods of survey data represented by a limited number of "point-in-time" counts. Nevertheless, in the case of diving ducks, the between-year differences large enough to suggest use of the Playgreen Lake segment of the Outlet Lake system by divers has declined following regulation.

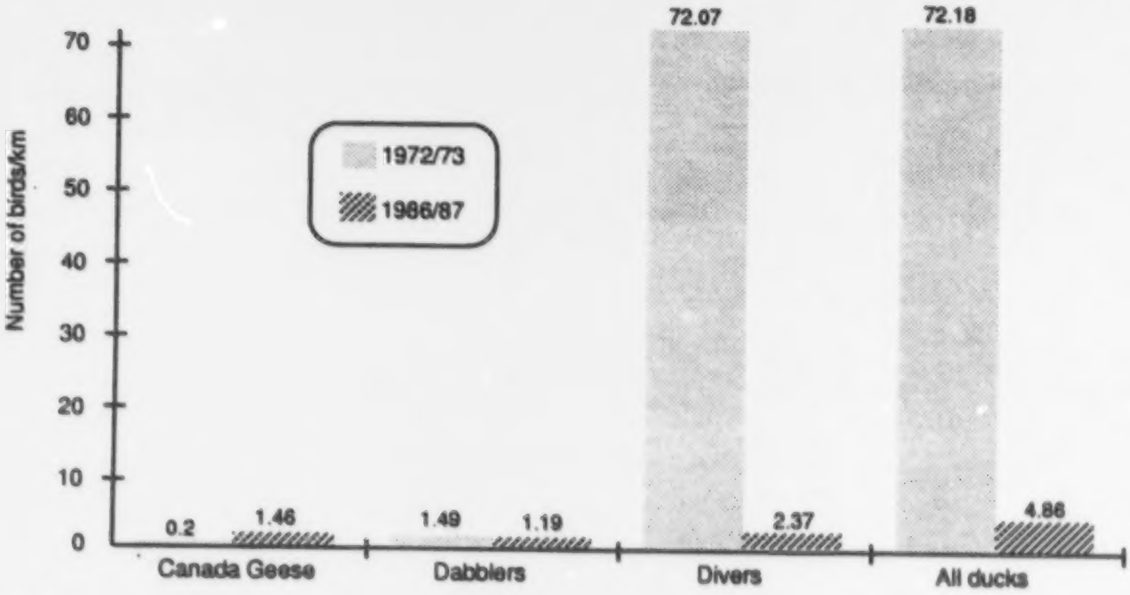

Figure 9. Pre- and post-regulation use of Playgreen–South Basin area by waterfowl based on highest survey densities.

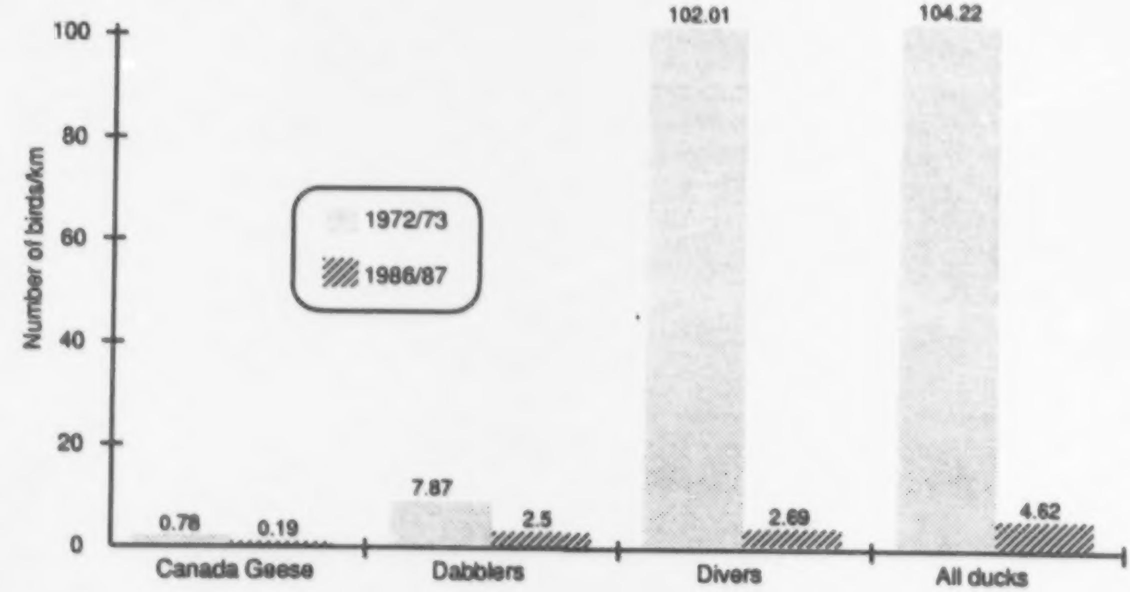

Figure 10. Pre- and post-regulation use of Playgreen–North Basin area by waterfowl based on highest survey densities.

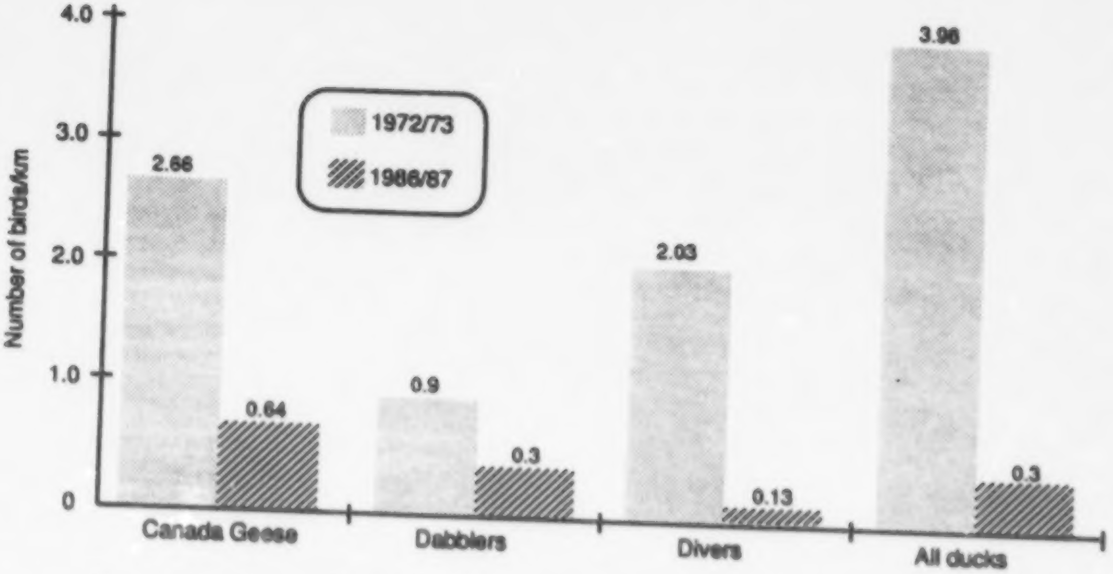

Figure 11. Pre- and post-regulation use of Kiskitto Lake by waterfowl based on highest survey densities.

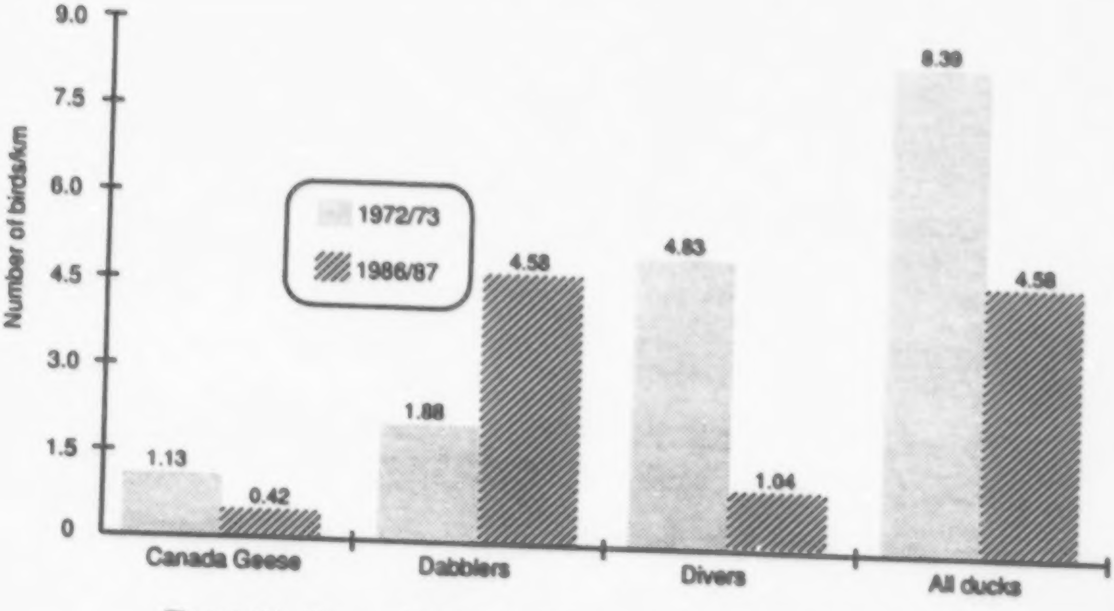

Figure 12. Pre- and post-regulation use of Kiskittogisu Lake by waterfowl based on highest survey densities.

Table 11. Changes in waterfowl densities* in the Outlet Lakes area from pre-regulation to post-regulation based on highest survey densities.

Species by Area	Pre-regulation 1972/73	Post-regulation 1986/87	% Change
PLAYGREEN LAKE — NORTH			
Divers	102.01	2.69	-97.4
All ducks*	104.22	4.62	-95.6
PLAYGREEN LAKE — SOUTH			
Divers	72.07	2.37	-96.7
All ducks*	72.18	4.86	-93.3
KISKITTO LAKE			
Divers	2.03	0.13	-93.6
All ducks*	3.98	0.3	-92.5
KISKITTOGISU LAKE			
Divers	4.83	1.04	-78.5
All ducks**	8.39	4.58	-45.4

* dabbling ducks and Canada geese have been omitted because of gaps and inconsistencies in survey counts

** includes dabblers, divers, & unidentified ducks.

Table 12. Summary of changes in waterfowl use of the Outlet Lakes area from pre-regulation to post-regulation based on highest survey densities.

Species	Pre-diversion 1972/73	Post-diversion 1986/87	% Change
Divers	41.85	1.57	-96.2
All ducks*	43.51	4.31	-90.1

* includes dabblers, divers & unidentified ducks.

7.2 Possible Causes of Waterfowl Use Changes

The following possible causes for the observed changes in diving duck numbers in the Outlet Lakes include:

1) Regional changes in waterfowl populations;
2) Modifications to the physical environment associated with Lake Winnipeg regulation; and
3) Habitat changes associated with Lake Winnipeg regulation.

7.2.1 Regional Waterfowl Population Changes

Regional changes in waterfowl populations can result from overexploitation by hunters or as a result of a widespread reduction in the availability or quality of habitat. If waterfowl populations had declined throughout northern Manitoba, one would expect such declines to be represented in waterfowl numbers in more localized areas such as the Outlet Lakes.

To determine whether the observed reduction in waterfowl use of the Outlet Lakes area could be attributed to a more widespread, regional decline in waterfowl populations, data collected annually by the U.S. Fish and Wildlife Service during the breeding pair surveys in Stratum 24 in northern Manitoba (Figure 13), were examined. These data provide adjusted population estimates for ducks and indicated breeding pair numbers for Canada geese by stratum and serve as general indicators of population size and trends.

It is important to remember that these data estimate the size of breeding populations of ducks and geese and do not indicate the size of fall migrating flocks or the "fall flight". However, the data are still useful as broad indices of waterfowl population size and trends.

Data for 1964-75 were selected to describe regional waterfowl population levels before LWR was fully operational, while data for 1977-88 represent regional population levels following regulation. The estimates of dabbler and Canada goose numbers are provided for general reference and indicate that, while

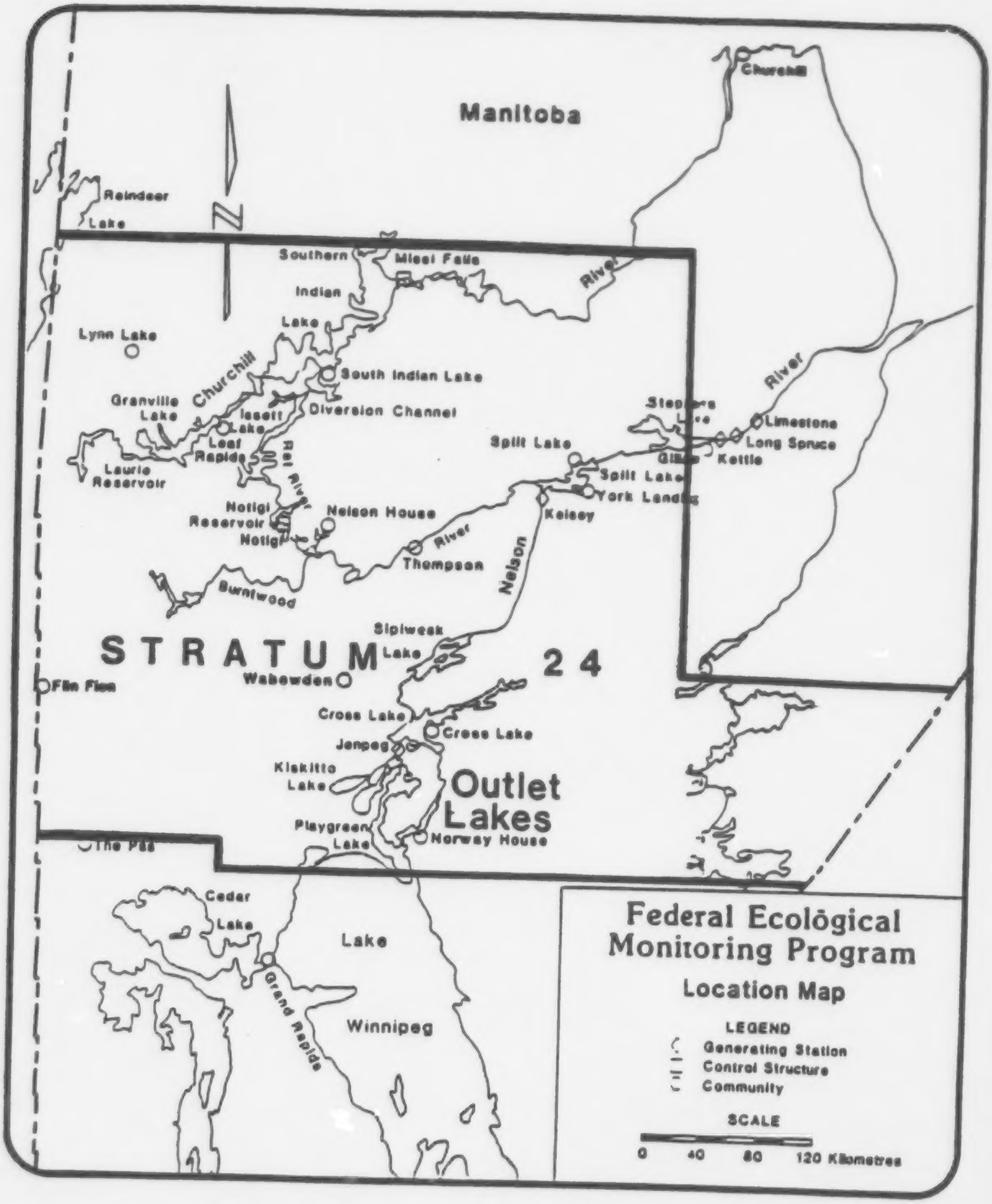
Manitoba
Churchill
N
Reindeer Lake
Southern Indian Lake
Missi Falls
River
Lynn Lake
Churchill
South Indian Lake
Granville Lake
Issett Lake
Diversion Channel
Leaf Rapids
Rat River
Laurie Reservoir
Notigi Reservoir
Notigi
Nelson House
River
Thompson
Burntwood
Nelson
Split Lake
Split Lake
York Landing
Kelsey
Stephens Lake
River
Limestone
Long Spruce
Gillam
Kettle
Sipiwesk Lake
STRATUM 24
Wabowden
Flin Flon
Cross Lake
Cross Lake
Jenpeg
Kiskitto Lake
Outlet Lakes
Playgreen Lake
Norway House
The Pas
Cedar Lake
Lake Winnipeg
Grand Rapids
Federal Ecological Monitoring Program
Location Map
LEGEND
Generating Station
Control Structure
Community
SCALE
0
40
80
120 Kilometres

Figure 13. Location of waterfowl survey Stratum 24.

populations have fluctuated considerably over the past 24 years, there is little evidence of either a sustained increase or decline in over time (Table 13). Diving duck population data for these two periods are also given in Table 13 and are discussed in greater detail below. Regional changes in diver populations that have occurred from 1955 to 1988 are displayed in Figure 14.

Figure 14 shows that large year-to-year fluctuations occurred in regional diving duck populations occurred during both the pre-and post-regulation periods. However, when the 12-year means for the two periods are compared, only a small change in numbers is evident from the pre-regulation period to the post-regulation period for divers (Table 14). Overall, the diver population estimate increased by 2.6 percent. Results of the 1973 and 1988 surveys for the Outlet Lakes area are included for comparison.

One must keep in mind, however, that the means on which these changes are based will vary somewhat as individual year's counts are added or subtracted. The main point to be made is that numbers of breeding waterfowl in the study region have neither declined nor increased over the long term, according to Stratum counts.

In conclusion, there is no indication that the observed reduction in waterfowl use in the Outlet Lakes area reflects a long-term decline in regional populations as represented by the Stratum 24 counts, since no such trends are evident in the stratum data. At the same time, one must be cautious about interpreting the significance of the 96 percent change in diver numbers derived from 1973 and 1988 surveys, because large between-year changes are not uncommon in waterfowl counts derived from aerial surveys.

For example, comparison of any pre-diversion year with any post-diversion year estimate using the Stratum 24 data can yield dramatically different results that may or may not have anything to do with changes in regional water conditions: comparison 1986 and 1970 Stratum counts suggest a 78 percent reduction in dabbling ducks, whereas comparison of 1978 and 1966 counts indicate a 437 percent increase. In the absence of counts compiled over several

Table 13. Adjusted population estimates (thousands) for dabblers, divers, and Canada geese in stratum 24 during pre-regulation (1964-75) and post-regulation (1977-88) periods (from U.S. Fish & Wildlife Service).

PRE-DIVERSION	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	MEAN
dabblers	228.9	404.4	205.0	716.0	1122.9	1102.3	1458.7	631.2	440.2	500.3	345.1	358.5	626.1
divers	198.2	276.7	228.3	233.0	587.7	653.3	788.0	447.4	513.8	375.5	270.2	383.2	412.9
Totals	**427.1**	**681.1**	**433.3**	**949.0**	**1710.6**	**1755.6**	**2246.7**	**1078.6**	**954.0**	**875.8**	**615.3**	**741.7**	**1039.1**
Canada geese	4560	6952	5145	5050	9504	15606	5925	4700	2498	1845	1590	2041.0	5451.3

POST-DIVERSION	1977	1978	1979	1980	1981	1982	1983	1983	1985	1986	1987	1988	MEAN
dabblers	708.9	1101.5	935.5	799.8	837.2	561.6	584.4	347.2	372.6	248.6	441.3	588.6	627.3
divers	536.2	376.2	336.9	387.6	746.3	329.4	531.7	319.4	359.1	534.1	339.8	288.2	423.7
Totals	**1245.1**	**1477.7**	**1272.4**	**1187.4**	**1583.5**	**891.0**	**1116.1**	**666.6**	**731.7**	**782.7**	**781.1**	**876.8**	**1051.0**
Canada geese	4444	6750	9743	5125	5681	2840	5000	2525	5517	1914	5345	8104.0	5249.0

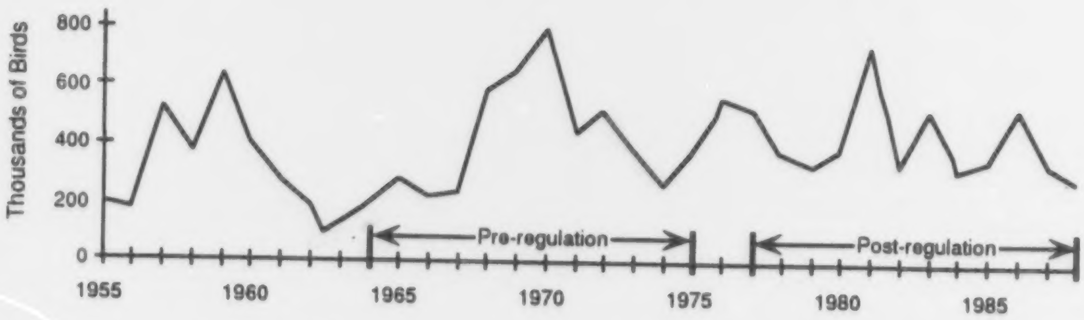

Figure 14. Adjusted population estimates of diving ducks in stratum 24 (northern Manitoba), 1955-1988.

Table 14. Changes in regional (Stratum 24) and local (Outlet Lakes) waterfowl use from pre-regulation to post-regulation.

Waterfowl	Stratum 24: 12-year Means*			Rat-Burntwood Counts**		
	1965-75 (pre)	1978-88 (post)	% Change	1973 (pre)	1988 (post)	% Change
Divers	412.9	423.7	2.6	41.85	1.57	-96.2

* adjusted population estimates (thousands)
** numbers represent birds/km

years, one would be hard-pressed to draw meaningful conclusions from these point-in-time data.

7.2.2 Transmission Lines and Channels

Norway House residents suggested that the transmission line constructed to provide electrical service to Norway House may have caused ducks to alter their flight patterns during fall migration. The transmission line spans both 2-Mile Channel and the natural outlet of Lake Winnipeg near Warren Landing. In addition, 2-Mile Channel itself represents an alteration to the physical environment.

As stated in Section 7.3, hunters from Norway House observed migrating flocks of ducks turn towards Lake Winnipeg at 2-Mile Channel as they crossed the south basin of Playgreen Lake in the fall, rather than using their traditional route past Warren Landing. Since the transmission line crosses both routes, it would appear that this change in flight pattern is more likely to be a result of new navigational cues associated with the new channel, and the shorter access to Lake Winnipeg, rather than avoidance of the transmission line in the Warren Landing area.

According to Manitoba Hydro records (D. Windsor, pers. comm.), the main transmission line to Norway House was in place by the end of the 1972-73 winter construction season (a branch line to Warren Landing was installed during the winter of 1973-74). Consequently, the line was already in place when four of the five pre-regulation aerial waterfowl surveys were conducted in the fall of 1973 (the fifth survey was conducted in the fall of 1972). If the presence of the transmission line caused ducks to avoid the Warren Landing area, one would expect that this effect would have already occurred in the fall of 1973 and would be reflected in the 1973 survey results. It is unlikely, then, that the observed reduction in duck numbers is related to presence of the transmission line.

If the presence of the transmission line and/or 2-Mile Channel was influencing waterfowl use or flight patterns, the effect would likely be localized.

The observed large reduction in diving duck use of the north basin of Playgreen Lake (Figure 11) cannot be attributed to localized changes in physical environmental conditions, such as 2-Mile Channel, which may be affecting local flight patterns.

In conclusion, it is unlikely that waterfowl use of the Outlet Lakes area has been affected directly by the presence of the transmission line or 2-Mile Channel. Consequently, it would not be valid to attribute the decline in waterfowl use to the physical presence of these features.

7.2.3 Habitat Changes

It is possible that the altered water regime, associated with Lake Winnipeg regulation, may have resulted in deterioration of waterfowl feeding and staging habitat and, consequently, reduced the attractiveness of the Outlet Lakes area to waterfowl during the fall staging and migration periods. It is also possible that 2-Mile Channel may have altered the quality of waterfowl habitat.

Increased water levels in Playgreen Lake levels apparently have caused flooding of shoreline vegetation and soils, resulting in increased amounts of sediment entering the Lake and remaining in the water column. MacLaren PlanSearch Inc. (1985) documented the susceptibility of sections of Playgreen Lake south basin shoreline to erosion and referred to the remnants of an old hydro pole situated in the water about 6 m from the southern shoreline in 1984. According to Manitoba Hydro records, the pole had been installed on land in 1972, some 15 to 30 m inland from shore. Based on this information, MacLaren PlanSearch estimated that between 21 and 36 metres of shoreline had eroded during the 12-year period between 1972 and 1984. Increases in turbidity, total organic carbon and total inorganic carbon, have been measured in the Norway House area following LWR and may be attributed to shoreline erosion or construction of 2-Mile Channel (Playle and Williamson 1986).

Water level regimes before and after Lake Winnipeg regulation in Playgreen Lake were examined using the data collected at Water Survey of Canada's Nelson River at Norway House station (#05UB001). The historic record (1915-87) was divided into pre-regulation (1915-74) and post-regulation (1977-87) periods. Data for 1975 and 1976 were excluded because they were considered to be transitional between pre- and post-regulation conditions. Excavation of 8-Mile Channel was not finished until September 13, 1975, with dredging completed on October 6, 1975. Similarly, 2-Mile Channel did not connect Lake Winnipeg to Playgreen Lake until May 12, 1976 and dredging was not completed until October 5, 1976.

Analysis of mean monthly water levels at Playgreen Lake during the pre-regulation and post-regulation periods suggest that mean post-regulation water levels were 8 to 23 cm higher than mean pre-regulation levels (Table 15), with the greatest increase occurring in September. Because the mean monthly values from both pre- and post-regulation periods have not been adjusted for variance within or between months and years, the significance of the monthly differences shown in Table 15 cannot not determined. For example, the range of variation reported from daily records for August, September, and October 1986 was 23, 53, and 41 cm, respectively, at Playgreen Lake. Similarly, comparison of mean monthly water levels from two sample years before regulation (1972/73) ranged from 2 to 67 cm; mean monthly differences for two post-regulation years (1986/87) ranged from 1 to 34 cm.

Comparison of pre- and post-regulation figures do indicate higher water levels prevailed during all months following regulation, which would be expected to alter shoreline conditions, including aquatic and near-shore terrestrial habitats. Therefore, it is reasonable to hypothesize that the observed changes in diving duck use of the Outlet Lakes area is associated with habitat changes resulting from the altered flow regime (i.e., slightly increased water levels on a year-round basis) following Lake Winnipeg Regulation.

That increased water levels and associated habitat change can affect use by diving ducks has been documented by Paloumpis and Starrett (1960) and Mills *et al.* (1966) along Illinois River.

Aside from the obvious ecological differences between aquatic systems in Illinois and northern Manitoba, the Illinois River and LWR projects share certain key characteristics. In both, the natural flow regime has been modified by man's activities, and in both cases, diving ducks have experienced the largest reduction in numbers.

Researchers have linked the tremendous decline in the number of lesser scaup using the Illinois River in the fall to the disappearance of fingernail clams (Sphaeridae) and small snails (Gastropoda) which were the main food items of lesser scaup at that location (Anders 1959, Rogers and Korschgen 1966). Increased turbidity and siltation and a substantial reduction in the abundance of aquatic plants have also been documented through various studies (Mills et al. 1966).

Table 15. Pre- and post-regulation water levels (m asl) at Playgreen Lake-Norway House hydrometric station (#05UB001).

	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
LONG-TERM MEANS												
1978-88 (Post)	217.04	217.03	217.04	217.14	217.29	217.47	217.49	217.49	217.38	217.27	217.12	217.02
1969-75 (Pre)	216.92	216.89	216.88	216.97	217.10	217.29	217.33	217.30	217.15	217.09	217.02	216.94
Difference*	0.12	0.11	0.16	0.17	0.19	0.18	0.16	0.19	0.23	0.18	0.10	0.08

* Given the degree of variation in water levels recorded both within months and between years, pre- and post-regulation differences shown here may not be statistically significant.

The fact that dramatic reductions in diving duck use accompanied alterations to the natural flow of water in both areas prompted a more detailed evaluation of waterfowl response to changes in water regime. Findings are presented in the following section.

7.3 Assessment of Habitat Change Impacts on Waterfowl Use

7.3.1 Methodology

To determine how the altered flow regime could affect waterfowl use of the Outlet Lakes area, a Valued Ecosystem Components (VECs) approach (Beanlands and Duinker 1983) was used to analyze those components of the environment having particular relevance to the specific assessment being conducted. The approach was used successfully in evaluating the potential environmental impacts of hydrocarbon development in the Beaufort Sea through the Beaufort Environmental Monitoring Project (Indian and Northern Affairs Canada 1983). More recently, the VEC concept was used to develop a screening system for identifying potential environmental concerns associated with Public Work Canada dredging proposals (Vonk et al. 1989).

The approach consisted of the following steps:

a) identification of the VECs;
b) identification of impact hypotheses that relate changes to the physical environment to the VECs; and
c) evaluation of impact hypotheses to determine validity.

7.3.2 Valued Ecosystem Components

First, the scientific literature dealing with the feeding habits of ducks; the relationships between benthic invertebrates and aquatic macrophytes; and the effects of turbidity, siltation and water level changes on macrophytes and invertebrates was thoroughly reviewed and the following VECs identified:

a) one or more diving duck species present;

b) one or more benthic invertebrate groups (e.g. amphipods) present;

c) one or more submerged macrophyte species (e.g. pondweeds) present; and

d) one or more components representing habitat attractiveness to ducks.

Lesser scaup was selected as a VEC because it is one of the most common diving duck species found in the Outlet Lakes area, and its food habits are well documented in the literature.

Table 16 summarizes the results of lesser scaup food selection studies. The preference towards animal foods represented by benthic invertebrates is immediately obvious. Of the invertebrates consumed, Gastropoda (snails), Pelecypoda (particularly Sphaeriidae, the fingernail clams), Amphipoda (amphipods) and Trichoptera (particularly caddis fly larvae) emerge as important food items. Due to the variety of ecological niches occupied by these different invertebrate types, "benthic invertebrates" were identified collectively as being a VEC, with each of the four groups representing "sub-VECs".

The existence of pondweed beds in Playgreen Lake is well known. MacLaren Plansearch Inc. (1985) described the distribution and abundance of *Potomogeton vaginatus* and *P. richardsonii* along the southern shoreline of the south basin of Playgreen Lake. The authors observed that *P. richardsonii typically* grows in shallower littoral zones, while *P. vaginatus* tends to inhabit deeper water. They also noted that the production of *P. richardsonii* may have diminished as a result of increased Lake levels associated with Lake Winnipeg regulation. For this reason, "pondweeds" were identified collectively as a VEC, with *P. richardsonii* and *P. vaginatus* as sub-VECs.

The means by which waterfowl navigate, recognize geographical landmarks and select habitats to meet their needs is not well-understood (e.g. Bellrose 1972, Wiltschko and Wiltschko 1978). This lack of understanding is not limited to waterfowl but, applies to all birds. Notwithstanding this knowledge deficiency, it is evident that waterfowl and other bird species do possess the ability to

Table 16. Food items (by % of diet) consumed by lesser scaup in Alberta, Saskatchewan, Manitoba, Northwest Territories, Minnesota, and Illinois.

Food Items	Sugden (1973), Alberta *	Dirschl (1969), Saskatchewan**	Bartonek & Hickey (1969), Manitoba^	Bartonek & Murdy (1969), N.W.T*	Afton et al. (press) Minnesota**	Anderson (1959) Illinois **	Rogers & Korschgen (1966), Illinois**
PLANT MATERIAL							
Carex spp.		0.4					
Lemna		0.5			0.3		
Myriophyllum spp.	2.0	0.3	5.0				
Scirpus spp.		2.2	7.0		0.2		2.9
Potamogeton spp.		3.3			0.1	2.8	3.3
Ceratophyllum demersum						2.4	
Misc. plants	2.0		2.0	1.0	6.6	4.5	0.3
Total Plants	**4.0**	**6.7**	**14.0**	**1.0**	**7.2**	**9.7**	**6.5**
ANIMAL MATERIAL							
Mollusca							
Gastropoda	16.0		3.0	1.0	3.0	43.4	70.1
Pelecypoda							
Sphaeriidae						32.3	11.9
Unionidae						2.6	2.9
Unidentified						0.3	0.1
Misc. Mollusca						7.6	
Total Mollusca	**16.0**	**0.0**	**3.0**	**1.0**	**3.0**	**86.2**	**85.0**
Arthropoda							
Crustacea							
Amphipoda	52.0	85.0	7.0	57.0	74.5		
Misc. Crustacea	1.0			2.0	2.9		0.5
Total Crustacea	**53.0**	**85.0**	**7.0**	**59.0**	**77.4**	**0.0**	**0.5**
Insecta							
Ephemeroptera						3.3	7.8
Odonata	3.0			17.0	0.2	0.2	
Hemioptera	3.0			11.0		0.3	
Coleoptera	1.0		2.0	4.0		0.1	
Diptera	16.0	0.3	12.0	1.0	8.9	0.1	
Trichoptera	2.0	4.5	24.0	6.0	1.9		
Misc. Insecta	1.0		1.0		0.2		0.2
Total Insecta	**26.0**	**4.8**	**39.0**	**39.0**	**11.2**	**4.0**	**8.0**
Hirudinae	1.0	3.5	31.0		1.2		
Misc. Animal			6.0				
Total Animals	**96.0**	**93.3**	**86.0**	**99.0**	**92.8**	**90.2**	**93.5**

**fall, both sexes; ^summer, males only; *late summer, both sexes recognize habitat types capable of fulfilling their needs and to distinguish good-quality habitat from poor-quality or unsuitable habitat (Hilden 1965). For this reason, a VEC representing a set of visual cues, related to landscape or habitat appearance, was identified. This VEC was termed "landscape characteristics".

In summary, four VECs were identified to represent "cause-and-effect" hypotheses:

a) lesser scaup;
b) benthic invertebrates;
c) pondweeds; and
d) landscape characteristics.

Four sub-VECs under "benthic invertebrates" are:

i) gastropods;
ii) amphipods;
iii) fingernail clams; and
iv) caddisfly larvae.

There are two sub-VECs under "pondweeds":

i) *P. richardsonii*; and
ii) *P. vaginatus*.

No sub-VECs were identified for the VEC "landscape" characteristics.

7.3.3 Impact Hypotheses

Figure 15 illustrates the possible linkages between increased water levels and reduced waterfowl use (diving duck use) of Playgreen Lake. Each linkage has been numbered for ease of reference.

An "impact hypothesis" can be defined as a statement that links a development-related activity to an environmental effect. In this case, the development-related activity is an increase in water levels, related to regulation of Lake Winnipeg, and the environmental effect is reduced diving duck use. Examination of the network diagram in Figure 15 indicates that two distinct "impact hypotheses" can be constructed to explain the relationship between water levels and waterfowl (diving duck) use:

A. **Increased water levels, resulting from regulation of Lake Winnipeg, have caused a reduction in the availability of invertebrate food items and, consequently, a reduction in the attractiveness of Playgreen Lake to diving ducks.**

B. **Increased water levels, resulting from regulation of Lake Winnipeg, have modified the landscape and habitat characteristics and, consequently, have reduced the attractiveness of Playgreen Lake to diving ducks.**

7.3.3.1 Hypothesis A

To determine whether an increase in the level of Playgreen Lake could have resulted in a reduction in the availability of invertebrates as food for diving ducks, it is necessary to investigate some of the factors which control or influence the abundance of invertebrates in aquatic ecosystems.

Potamogeton richardsonii favours the shallower littoral zones of Playgreen Lake while *P. vaginatus* tends to inhabit the zones of deeper water. MacLaren Plansearch Inc. (1985) acknowledged that *P. richardsonii* may have diminished as a result of increased lake levels associated with Lake Winnipeg regulation. In fact, it is likely that increased lake levels have caused stands of *P. vaginatus* to become more widespread at the expense of *P. richardsonii*. Robel (1962) documented the sensitivity of *P. pectinatus* to water level changes. A rise in water level of less than 8 cm caused vegetation production to increase by 32 percent in shallow areas and decrease by 35 percent in the deeper areas.

The association of high densities and diversity of invertebrates with aquatic macrophytes, including pondweeds such as *P. vaginatus* and *P. richardsonii*, is well documented in the literature. Researchers have noted that benthic invertebrates inhabiting lake substrates, such as snails and fingernail clams, are much more abundant in vegetated areas than in non-vegetated areas (e.g. Krull 1970, Engel 1988). Voigts (1976) found that amphipods were most abundant in dense beds of submerged vegetation.

Many benthic invertebrate species depend on aquatic macrophytes to meet essential life cycle requirements. The leaves of aquatic plants provide a substrate for egg deposition by several invertebrate species, including snails (Gerrish and Bristow 1979, Pandit 1984). Macrophytes support large quantities of periphyton (a film of microorganisms, chiefly bacteria, protozoa and algae), a favoured food source for amphipods (Brown et al. 1988) and snails (Pandit 1984). Caddisfly larvae use the leaves of submerged macrophytes for building dwelling cases (Soszka 1975). Biggs and Malthus (1982) noted small bivalve mussels, family Sphaeriidae, were associated with the roots of all macrophytes.

Some submerged aquatic plants provide habitat for a greater diversity and number of invertebrates than others. Gerrish and Bristow (1979) concluded that growth form and leaf morphology of aquatic plants are the main factors determining the size of the resident macroinvertebrate population. Dvorak (1987) noted that, despite their lower biomass, submerged plants offer much larger surface area for macroinvertebrate colonization than emergent plants, which have a much smaller surface area in relation to their higher biomass.

The importance of surface area of submerged aquatic plants in determining the number and diversity of invertebrates present was noted by Krecker (1939), who observed that the pondweed *P. crispus*, which has crenulate leaves, supported more than twice as many invertebrate genera and numbers as *P. compressus* and *P. pectinatus*, which have narrow, thread-like leaves.

Berg (1949) studied insects and insect parasites associated with 17 species of *Potamogeton*. For the Outlet Lakes study, the 17 species were classified into broad-leaved and narrow-leaved group. Table 17 shows that more insect species

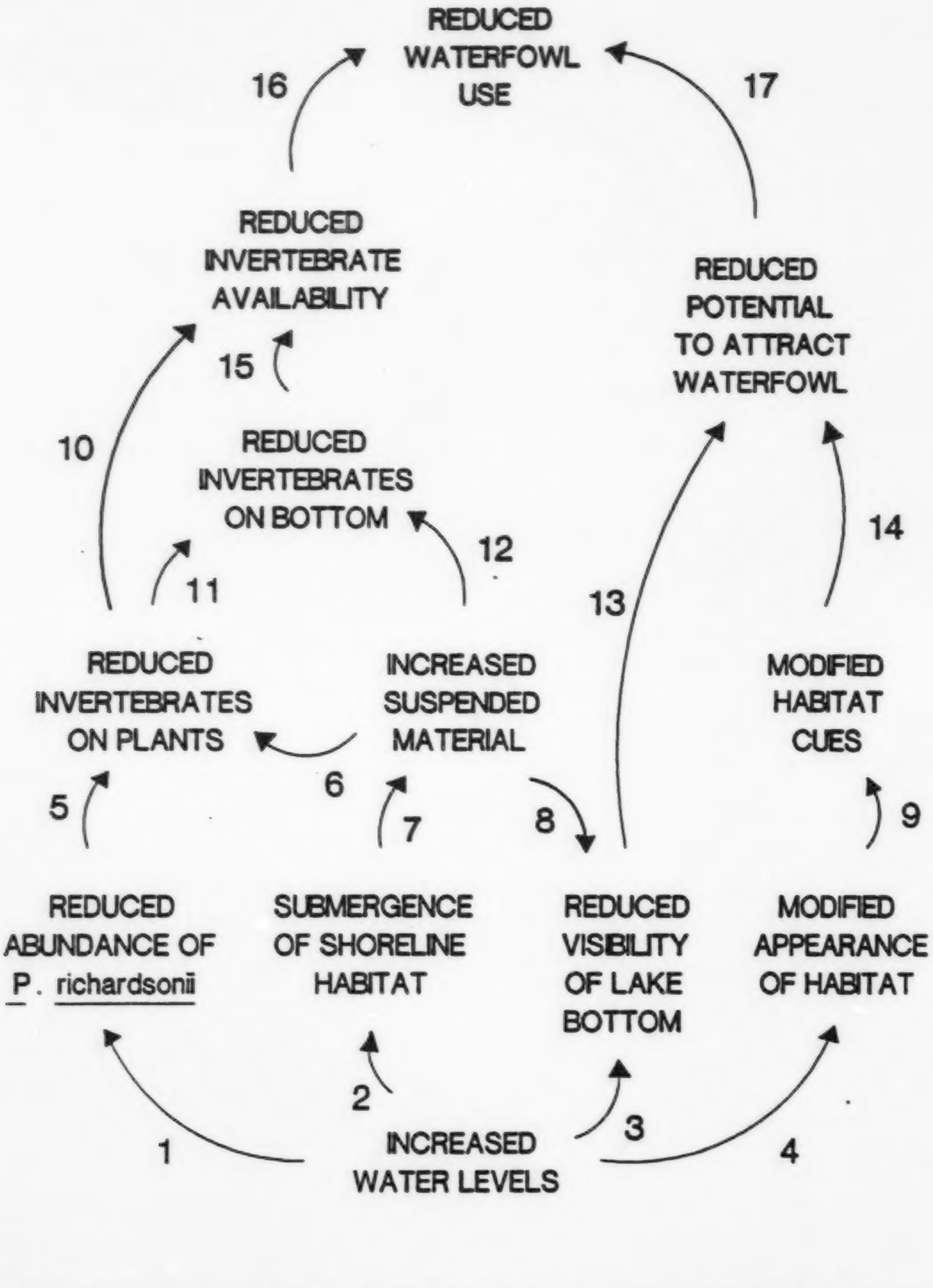
REDUCED WATERFOWL USE
16
17
REDUCED INVERTEBRATE AVAILABILITY
REDUCED POTENTIAL TO ATTRACT WATERFOWL
15
10
REDUCED INVERTEBRATES ON BOTTOM
12
14
11
13
REDUCED INVERTEBRATES ON PLANTS
INCREASED SUSPENDED MATERIAL
MODIFIED HABITAT CUES
6
5
7
8
9
REDUCED ABUNDANCE OF P. richardsonii
SUBMERGENCE OF SHORELINE HABITAT
REDUCED VISIBILITY OF LAKE BOTTOM
MODIFIED APPEARANCE OF HABITAT
2
3
1
4
INCREASED WATER LEVELS

Figure 15. Network diagram showing relationship between waterfowl use and various environmental factors in the Playgreen Lake area.

were found on broad-leaved species than on narrow-leaved *Potamogeton*. Berg pointed out that caddisfly larvae can obtain more abundant and better materials for casemaking from broad-leaved aquatic plants than from narrow-leaved plants.

It was noted earlier that increased lake levels would cause stands of *P. richardsonii* to decline (linkage 1 in Figure 15) and *P. varnatus* to increase. Since *P. richardsonii* is a broad-leaved pondweed, while *P. vaginatus* is narrow-leaved, a reduction in the distribution and abundance of *P. richardsonii* would lead to an overall reduction in the abundance [illegible]ertebrates (linkage 5). Several aquatic invertebrate groups (e.g. amphipods, snails) spend part of their lives on the submerged plant substrate, but also inhabit the bottom substrate where plants are growing. Therefore, a reduction in invertebrate numbers on plants could be accompanied by a reduction in invertebrate numbers on the bottom substrate (linkage 11), although the existence of such a cause-effect relationship would have to be verified through field studies. The combined effect would be a reduction in the availability of invertebrates as food for diving ducks (linkages 10 and 15). With less animal food available, Playgreen Lake would be less attractive to diving ducks and waterfowl use would be expected to decline (linkage 16).

Flooding and submergence of shoreline vegetation and soils (linkage 2) due to increased water levels is expected to increase the amount of sediment entering Playgreen Lake and remaining in the water column (linkage 7). It should be noted that the 0.75 to 0.78 m increase in water depth at Playgreen Lake is lower than historic high water levels recorded before LWR. Nevertheless, water levels will remain high for long periods under the regulated regime; this, coupled with the reversal of the pre-regulation flood-drawdown cycle, has likely affected shoreline erosion patterns and growth of aquatic vegetation.

Increased levels of sediment adversely affect aquatic invertebrates but benefit others, both directly (linkage 12) and indirectly through effects on aquatic macrophytes (linkage 6).

Ellis (1936) observed that most of the common freshwater mussels (Mollusca) were unable to maintain themselves on sand or gravel bottoms (normally satisfactory bottom habitat) that were blanketed with silt, even when silt

Table 17. Numbers of insect species observed on broad-leaved and narrow-leaved pondweeds (Berg 1949).

Pondweed Species	No. of Insect Species (including parasites)	No. of Insect Species (no parasites)
BROAD-LEAVED		
P. alpinus	13	8
P. amplifolius	25	20
P. gramineus	10	8
P. illinoensis	5	5
P. natans	23	16
P. nodosus	6	5
P. oakesianus	4	1
P. praelongus	10	7
P. richardsonii	18	16
MEAN	**12.7**	**9.6**
NARROW-LEAVED		
P. epihydrus	6	4
P. filiformis	0	0
P. foliosus	2	2
P. friesii	1	0
P. pectinatus	1	1
p. pusillus	0	0
P. robbinsii	6	6
P. zosteriformis	7	4
MEAN	**2.9**	**2.1**

deposition was less than 3 cm deep. High mussel mortality was found to be induced by the silt covering rather than low oxygen, pH, or other water quality characteristics. Laboratory experiments showed that silt interfered with the feeding of freshwater mussels.

Siltation, or sedimentation, is believed to have been a major factor in the almost complete disappearance of pondweeds and other large aquatic plants along the Illinois River in Illinois (Mills et al. 1966). The authors state that "siltation affects aquatic plants adversely in two ways: it produces a turbidity which reduces the penetration of light and inhibits photosynthesis, and it creates bottom conditions which make it difficult or impossible for various species of plants to obtain anchorage when they are buffeted by wave action". The great reduction in aquatic plants was accompanied by virtual disappearance of snails and fingernail clams. Siltation was considered to be an important factor influencing the drastic reduction in bottom fauna (Paloumpis and Starrett 1960).

Paloumpis and Starrett (1960) attributed the scarcity of lesser scaup at Quiver Lake, on the Illinois River, to the disappearance of small snails and the decline of fingernail clams. Mills et al. (1966) similarly linked the major decline in lesser scaup along Illinois River in fall to disappearance of fingernail clams, citing Anderson's (1959) findings that mollusca comprised over 85 percent of the diet of lesser scaup (Table 16).

Increased sedimentation and turbidity, combined with increased water depth resulting from a rise in Lake level, would cause a reduction in visibility and could interfere with the ability of diving ducks to locate food items in the water column and on the bottom of Playgreen Lake (linkages 3 and 8).

In a controlled-environment study using a large aquarium, Tome and Wrubleski (1988) noted that lesser scaup appeared to locate prey visually. The authors observed that diving scaup frequently stopped and directed foraging movements of the bill towards amphipods swimming in the water column. They consumed prey distributed on artificial vegetation placed in the aquarium and often pursued and consumed amphipods that would drop to the substrate when the 'vegetation' was disturbed. Increased turbidity combined with increased depth would tend to reduce visibility, and would likely make such waters less attractive to diving ducks.

Results of surveys conducted by Fisheries and Oceans Canada (DFO) in 1987 and 1989 indicated that standing stocks of various benthic invertebrate groups inhabited the south and north basins of Playgreen Lake and in Kiskittogisu Lake (McKerness 1990). However, these results do not necessarily mean that those invertebrates were available to diving ducks such as scaup. Benthic invertebrate sampling stations were located on open water transects which did not include the near-shore vegetated zones of the lakes sampled (D. Rosenberg, pers. comm.). Therefore, while the margins of the diving duck feeding areas were not covered by the DFO surveys, open-water areas typically used by diving ducks for foraging were sampled. In addition, the DFO surveys were both conducted in July. No data was collected on the abundance of benthic invertebrates during the fall waterfowl staging period of September and October. Consequently, no firm conclusions can be drawn regarding the abundance or availability of benthic invertebrates to fall-staging diving ducks such as scaup.

The above analysis suggests Impact Hypothesis A (that increased water levels, associated with Lake Winnipeg regulation, may have reduced the availability of invertebrate food items to diving ducks, and contributed a decline in diving duck use of the Outlet Lakes area) has some validity, although field studies are required for verification.

7.3.3.2 Hypothesis B

Hypothesis B assumes that increased water levels have modified the landscape and habitat characteristics of the Outlet Lakes area, thereby reducing the area's attractiveness for diving ducks.

Although many scientific papers have been published on the topics of bird navigation and habitat selection, the level of understanding of the actual mechanics involved remains poor (Bellrose 1976). Hilden (1965) in his review of habitat selection mechanisms in birds during the breeding season, refers to "proximate" and "ultimate" factors which operate together and result in the selection of habitat.

Ultimate factors are essential for the survival of a species and include: food, requirements imposed by structural and functional characteristics of the species, and shelter from enemies and adverse weather. Proximate factors are certain characteristic stimuli of the species-specific habitat that release what is referred to as the "settling reaction". These factors include: landscape, terrain, feeding and drinking sites and other animals.

Not every suitable habitat need possess all the features characteristic of the optimal environment. Rather, it appears the positive and negative habitat characteristics are "summed" and, when the combined effect of the stimuli exceeds the threshold of the settling reaction, the habitat is selected. Similarly, one key stimulus, such as adverse landscape appearance, may out-weigh the other stimuli; in that situation, the combined effect of other positive stimuli would not be sufficient to induce the bird to "settle" in that particular habitat. Studies on wild geese conducted by Barry (1962) and Wood (1964) concluded that, where certain proximate factors such as appropriate nesting habitat conditions are lacking, ovulation can be inhibited and the nesting cycle terminated. Marshall (1952) speculated that inhibitions to breeding in birds appear to be brought about by the lack of one or more important environmental stimuli necessary to complete the reproductive cycle.

Habitat selection is probably a two-stage process (Hilden 1965). The first stage, settling down and exploring a particular habitat area, is released by features of the landscape and general characteristics of the terrain. Whether the habitat is suitable or not depends on how closely certain of its details conform to the other stimuli constituting the proximate factors in the bird's habitat selection mechanism.

Landscape and terrain are important "proximate" factors influencing habitat selection. Sometimes, the key factor for habitat selection may seem subtle but is still biologically important. For example, the lapwing (*Vanellus vanellus*), prefers grey-brown meadows and avoids green ones. The former are poor meadows where the grass will remain low, whereas the latter are fertile and the grass will later grow high. As the lapwing is adapted to live in low grass, this selection mechanism guides the bird to the most suitable environment. The colour of the meadow (a proximate factor) serves as a reliable indicator of a favourable breeding place (an ultimate factor).

At Playgreen Lake, there are a number of negative environmental stimuli, related to habitat quality, that may serve to repel diving ducks. Higher lake levels have modified the appearance of shoreline and off-shore habitat (linkage 4). The band of emergent vegetation along the shoreline has been decimated by deeper water. Turbidity levels in the Lake have increased which, in turn, has decreased visibility through the water column. In short, higher lake levels have modified (linkage 9) habitat characteristics (cues), thereby reducing the potential of the habitat to attract diving ducks (linkage 14).

While the reaction of birds to stimuli guiding them to suitable habitat is initially inherited, it is apparent that this innate behaviour is either reinforced or modified at a later stage by learning (Hilden 1965). Therefore, while Playgreen Lake used to attract reasonably large numbers of diving ducks, it appears that, at some point following Lake Winnipeg regulation, the ducks may have discovered that the quality of habitat had declined to a point where it did not meet their requirements for staging and, therefore, received less use during fall (linkage 17).

Conditions in the Outlet Lakes area following LWR tend to support the validity of Impact Hypothesis B: that increased water levels, associated with Lake Winnipeg regulation, have reduced the attractiveness of the Outlet Lakes area to diving ducks through modification of landscape and habitat characteristics.

8. SUMMARY AND CONCLUSIONS

8.1 Project Impacts on Waterfowl and Waterfowl Habitat

Comparison of waterfowl survey data collected in 1972/73 and 1986/87 suggests that diving duck use of the Outlet Lakes area in fall has declined following completion of the LWR Project in 1976. Results of pre- and post-regulation surveys were insufficient to estimate changes in levels of use by dabbling ducks and Canada geese with any degree of confidence.

Densities of diving ducks observed along shorelines of Outlet Lakes in the fall of 1986 and 1987 were approximately 96 percent lower than densities recorded prior to Lake Winnipeg regulation in 1972 and 1973. The largest reductions (97 percent) in diving duck use have occurred in the north and south basins of Playgreen Lake.

The observed decline in the use of the Outlet Lakes area by divers cannot be attributed to changes in regional waterfowl populations documented in northern Manitoba. There is no indication from Stratum 24 counts that regional diver populations in the study region have increased or declined in the long term, although their numbers have fluctuated dramatically from year to year.

Analysis of water level data for Playgreen Lake indicates that slight year-round increases in water levels following Lake Winnipeg regulation have altered the quality of Playgreen Lake habitat and may have reduced the attractiveness of Playgreen Lake to diving ducks in the fall.

8.2 Project Impacts on the Norway House Band

Many, if not most, of the preferred duck hunting areas traditionally used by Norway House Bank Indians are located in the south basin of Playgreen Lake, where the greatest reduction in diving duck use was observed. These areas include the eastern shoreline of the Lake south of Norway House, the narrows near Warren Landing, Kettle Island, and Sepastak Channel.

Norway House residents interviewed during the study indicated that reduction in waterfowl availability has limited opportunities for Norway House Band members to hunt and view waterfowl.

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APPENDIX A

NORTHERN FLOOD AGREEMENT

Background

The **Northern Flood Agreement** (NFA) was signed on December 16, 1977 by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee to establish a mechanism for compensating members of the Nelson House, Norway House, Cross Lake, Split Lake and York Landing (formerly York Factory) bands for adverse effects caused by the hydroelectric development.

The Preamble of the Agreement lays out the premises and principles which together provide a basis for the Agreement's provisions. Clauses C, D, E and G of the Preamble are particularly relevant to the present study and are reproduced below verbatim:

> *C.* *As a result of the modification of the water regime, adverse effects have occurred and may continue to occur, on the lands, pursuits, activities and lifestyles, of the residents, individually and collectively, of the Reserves of Cross Lake, Nelson House, Norway House, Split Lake and York Landing ("the Reserves" as hereinafter defined);*
>
> *D.* *The parties wish to ensure that all persons as defined herein, who may be, or have been, directly or indirectly, adversely affected by the* [Lake Winnipeg Regulation and Churchill River Diversion] *Project shall be dealt with fairly and equitably;*
>
> *E.* *Uncertainty as to the effects of the Project, with respect not only to the Project as it exists at the date of this Agreement but also as it may develop in he future, is such that it is not possible to foresee all the adverse results of the Project nor to determine all those persons who may be affected by it, and, therefore it is desirable to establish through the offices of a single arbitrator a continuing arbitration instrument, to which any person may submit a claim,*

and as well as to fully empower such arbitrator to fashion a just and appropriate remedy;

G. *Canada, by virtue of its jurisdiction and responsibility for Indians and land reserved for Indians, is committed to playing an active role in providing opportunity for the continued viability of the communities and, in particular but without limitation, in making available resources and expertise to the communities in planning and improving the social and economic conditions of the communities, and in ensuring that the special rights of Indians, including those arising from Treaty 5, are adequately protected.*

Contractual Obligations of the Northern Flood Agreement

The provisions of the NFA are arranged in a set of 23 Articles which represent contractual obligations. Three subarticles ultimately lead to the initiation of the present study and are, therefore, reproduced below verbatim:

17.1 *Hydro and Canada and Manitoba, severally and jointly, undertake to implement such recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board Report which affect the communities and which fall within their respective or joint jurisdictions.*

17.5 *In particular but without limitation, monitoring of adverse effects of the Project pursuant to the Lake Winnipeg, Churchill and Nelson Rivers Study Board recommendations shall be planned and implemented so as to provide information as may be necessary to give effect to this Agreement.*

18.2 *Canada and Manitoba recognize that the Project is intended to benefit all citizens of Canada, and most particularly of Manitoba, on the one hand, and that the resource users have been and may continue to be adversely affected on the other hand, and that it is in the public interest to ensure that any damage to the interests, opportunities, lifestyles and assets of those adversely affected be compensated appropriately and justly.*

Claim 18

In December 1981, the five NFA Indian Bands and the Northern Flood Committee filed a claim (Claim 18) alleging that Canada, Manitoba and Manitoba Hydro (the Respondents) had failed to implement recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board and to report to the NFA bands on implementation measures.

In particular, they claimed that the Respondents had "failed to establish a mechanism to deal with monitoring and analysis of ongoing social and economic changes related to hydro-electric development and more generally, northern development pursuant to Recommendation 5(c) of the Study Board". In addition, they claimed that the Respondents had "failed to develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the area affected by the Hydro Project pursuant to Recommendation 10 of the Study Board".

Failure to implement these two recommendations meant that the adverse ecological effects of the Project, and the associated adverse social and economic effects, had not been monitored and the information necessary to give effect to the NFA not provided. Consequently, the requirements of NFA sub-article 17.5 had not been met. Furthermore, without this information, it has not been possible to determine to what extent damages to the interests, opportunities, lifestyles and assets of resource users have occurred and, therefore, the level of compensation that is necessary (sub-article 18.2).

Filing of the claim resulted ultimately in approval of funds by Treasury Board on February 6, 1986, for implementation of the NFA over the period 1986-91 (T.B. Minute 800953). Availability of these funds permitted the Canadian Wildlife Service to initiate studies of the adverse effects of the project on waterfowl use of the Rat-Burntwood rivers system in the fall of 1988.